CONNECTIVITY RESTORATION FOR FISHES IN POST-INDUSTRIAL RIVERS OF NORTH EAST ENGLAND

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Abstract

Many rivers in developed regions experienced a strong decline in ecological function during the Industrial Revolution, due to poor water quality, degraded habitat and diminished hydrological connectivity. Post-industrially, water quality has dramatically improved in many rivers, and clean-water indicator species have returned, yet such rivers often remain very fragmented by river engineering, with locally degraded habitat and resultant effects on ecological communities, especially of fishes. River restoration activities are widespread, but their effectiveness in restoring biodiversity and ecological function remain poorly known. This study explores the causes of decline of fish populations in rivers of industrial North East England, their partial recovery, and the role of river restoration, especially through removal and mitigation of anthropogenic river barriers.

In a historical review of the decline and partial recovery of the rivers Tyne, Wear and Tees, and their fish stocks, it was found that before the 19th Century Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) were abundant in all three rivers. These catchments were subject to heavy industry and urbanization, instream barrier construction, and industrial pollution from the 19th Century to the mid-20th Century, during which time their fish stocks dramatically declined. Following decreased heavy industry, closure of mines and improvements in wastewater treatment, salmon and sea trout started to recover in the Tyne and Wear from the 1960s onwards and stabilized in recent years; these rivers are now the first and second best salmon rivers in England, in terms of angler catches. By comparison, anadromous salmonid numbers in the Tees increased much more slowly, potentially and partly due to impacts from the Tees Barrage. In general, the potential for recovery of anadromous salmonid stocks in post-industrial Pennine rivers appears driven by both accessibility and survival in the river, through effects of barriers, pollution and predators.

Since river reconnection programmes require barrier inventories for restoration planning,

the adequacy of the current national barrier inventory was assessed by field surveying two medium-sized catchments, the Wear and the Tees. The national river barrier inventory was found to be highly incomplete. From surveyed reaches across both catchments, 77.3% of barriers were found to be missing from the national database, including 68.6% of artificial barriers and 82.6% of natural barriers. Only 21.5% of artificial barriers had been removed or mitigated in both catchments, suggesting that river restoration in Northeast England, and perhaps in England more generally, still has a long way to go.

The effectiveness of barrier removal on habitat change and responses of fish and invertebrate communities was studied in a small stream joining the Tees estuary. Removal of a small tidal barrier increased habitat diversity immediately upstream, while changes in the invertebrate community up- and downstream were minor and transitory. A dramatic and sustained increase in fish density occurred in the previously impounded zone. The upstream recolonization of European eel (*Anguilla anguilla*) was greatly increased within two years. The eel density in the previously impounded zone increased from 0.5 per 100 m^2 before barrier removal to 32.5 per 100 m^2 five months after removal. In contrast, the population of brown/sea trout (*S. trutta*) has not yet benefitted from barrier removal, suggesting wider catchment management such as habitat and water quality improvements are required to complement connectivity restoration.

In rivers or tributaries with multiple barriers, catchment-scale connectivity restoration may be needed to help restore the density and distribution of diadromous and river-resident fish species. Extensive within-tributary fish sampling was used to determine local and sub-catchment responses to partial connectivity restoration. It was found that benefits of connectivity restoration in streams with many barriers may take several years to develop and that stochastic events on fish populations can obscure restoration responses. Compared with fish pass installation, barrier removal was found to be more effective in restoring lotic habitat and fish species, and facilitating movement of poorly dispersing species such as bullhead (*Cottus perifretum*). Findings of this thesis underline the importance of managing in-stream barriers sensitively, and have contributed to our understanding of the effects of connectivity restoration on post-industrial rivers.

Connectivity restoration for fishes in post-industrial rivers of North East England

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Declaration

The material contained within this thesis has not previously been submitted elsewhere for any other degree or qualification. The research reported here has been conducted by the author unless stated otherwise.

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Chapter One

General Introduction

1.1 Global overview

Worldwide, many river systems have been impacted by environmental degradation due to agriculture, industry, urbanization and related impacts (Jackson *et al.*, 2001; Naiman *et al.*, 2002; Nilsson *et al.*, 2005; Jansson *et al.*, 2007). Intensive human development in many river systems has caused a strong decline in ecological function, resulting from poor water quality, degraded habitat and diminished hydrological connectivity (Dudgeon *et al.*, 2006). Fish are one of the most important components of aquatic ecosystems. However, their abundance and diversity have been impacted by a series of problems such as habitat degradation, fragmentation, and destruction; environmental pollution; invasive species; overfishing; and climate change (Figure 1.1; Larinier, 2001; Naiman *et al.*, 2002; Nilsson *et al.*, 2005; Agostinho *et al.*, 2008). Rivers in the UK have a long history of anthropogenic modification (Figure 1.2; Lewin, 2013; Addy *et al.*, 2016). Over a thousand years ago in Europe (and longer ago in some civilizations) people farmed and built towns alongside rivers, altered river courses and diverted water to water wheels to power mills. The Industrial Revolution in Britain and many other countries speeded up the modification of river systems (Deane, 1980; Mawle and Milner, 2003). More recently, due to the fast-growing human population and the increasing demand for water, more and more human activities have impacted on rivers. Hydraulic structures such as dams have increased dramatically in the past 50 years, it is estimated there were nearly 5,000 large dams (>15 m height) all over the world by 1950, and the numbers increased to 45,000 by the year 2000 (Khagram, 2004; Nilsson *et al.*, 2005).

Before the 1980s, efforts to quantify human impacts on rivers, and to rehabilitate these, were dominated by water quality (especially chemical water quality), but increasing recognition of the multiplicity of factors degrading good ecological functionality has progressively resulted in a more integrated ecosystem approach (e.g. Water Framework Directive [WFD], 2003) (Reyjol *et al.*, 2014; Vlachopoulou *et al.*, 2014).

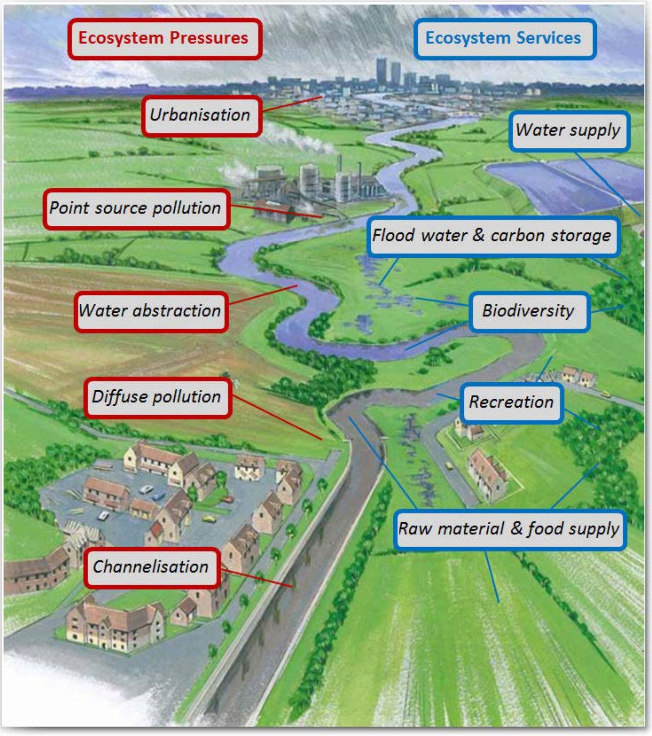


Figure 1.1 Illustration of some components of river degradation. Source from River Restoration Centre (2016). Note that some key elements such as dams and weirs are not included.

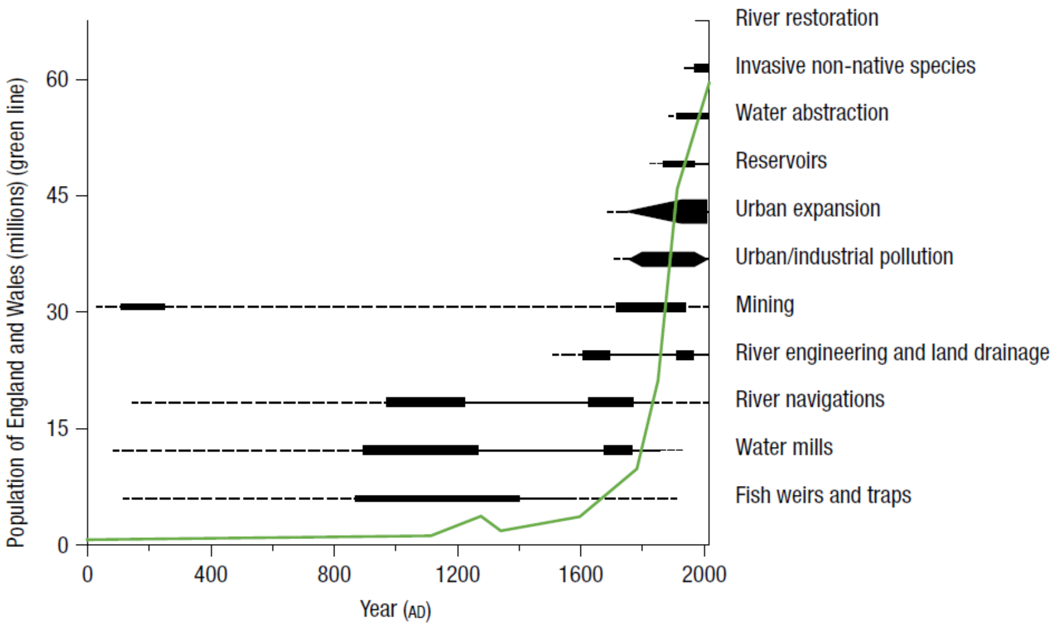


Figure 1.2 Timeline of key human actions that have altered rivers in England and Wales (Lewin, 2013; Addy *et al.*, 2016). The figure is missing some notable factors, such as diffuse pollution. The main impact periods for each are graded by line weightings. Solid line: proven evidence of human intervention. Dotted line: possible human intervention.

According to data obtained by River Habitat Surveys under the WFD, more than 50% of river length in England and Wales has been physically modified (Maltby *et al.*, 2011; Addy *et al.*, 2016). In Scotland, 17% of water bodies suffered from channel and bank modifications, and 16% of river length was impacted by barriers to fish migration (Maltby *et al.*, 2011). Rivers in urban areas are mainly impacted by bank reinforcement and re-sectioning (Boitsidis *et al.*, 2006), such as using concrete, gabions, boulders, soils or retaining walls to prevent river bank erosion or failure and to reduce channel movement. The impacts of bank reinforcement include stream channel narrowing, water velocity increases and uniformity, habitat alteration and reductions in fish populations (Schmetterling *et al.*, 2001).

Dam construction for water supply and hydroelectricity has been another important feature in the anthropogenic modification of rivers. The expansion of human populations and increased water usage has led to extensive damming of rivers and streams all over the world (Khagram, 2004). Dams interrupt water flow, cause hydrological changes and break river ecosystem continuity. During the 19th Century, large dam construction increased sharply from less than 10 to 175 in the UK and the construction rate grew from 1.7 to 5.4 dams per year after 1950 (European Environment Agency, 2008). There are now 486 large dams (> 15 m head) in the UK (European Environment Agency, 2008), but this number is dwarfed by there being more than 20,000 anthropogenic river obstacles in England and Wales alone (Entec, 2010; Jones *et al.*, 2019). Dams and other obstacles can prevent or delay fish migrations for reproduction or feeding (Lucas and Baras, 2001). Dams can cause changes in water temperatures, oxygen levels and sediment, redirect river channels, and disrupt river continuity. They can also have indirect effects, for example building a dam may increase predation on migratory fish delayed in passing the dam (Larinier, 2001). These problems can lead to the decline of fish populations or may promote the colonisation and persistence of non-native species (Jansson *et al.*, 2007).

Globally, 48% of the total volume of large rivers is either moderately or severely impacted by artificial flow regulation and/or fragmentation (Grill *et al.*, 2015). These impacts strongly affect the migratory patterns of fish and have caused damage to river habitats. A river that flows unaffected from its source to its mouth without encountering any dams, or other anthropogenic physical barriers is defined as a free-flowing river by the WWF (WWF, 2006). These rivers are crucial to migratory fish by providing a variety of habitats, flows and food sources (Brink *et al.*, 2018). However, the numbers of free-flowing rivers have progressively reduced in recent years and have become few and far between (Brink *et al.*, 2018). A study on river fragmentation has shown less than 40% of large rivers (> 1000 km) globally are still free-flowing, and most of these rivers are actually tributaries of major river systems in the world (WWF, 2006).

In order to restore the natural state and functioning of river systems, an increasing number of river restoration projects have been carried out, though usually only on small sections of river or stream (Palmer *et al.*, 2005; Jansson *et al.*, 2007; Nilsson *et al.*, 2015). European legislation (European Water Framework Directive, Habitats and Species Directive, Floods Directive) in the UK provides the main driver for river restoration, and this policy is expected to continue even after the UK has left the EU. It is estimated that more than 2600 river restoration projects have been undertaken in the UK since the 1980s (Addy *et al.*, 2016; River Restoration Centre, 2016). The aim of these projects is to recover the river habitat and restore river biodiversity, or at least reduce the rate of biodiversity loss. Although the number of river rehabilitation (restoration: aims for full recovery of natural processes, rehabilitation: recognizes that partial recovery is the maximum likely to be achievable, although these terms are often used interchangeably in the literature) schemes is increasing and widespread in the UK, the effects of these projects on river biota are usually poorly understood (Pretty *et al.*, 2003). There have been few holistic assessments to evaluate the effectiveness of the measures on ecological characteristics in these projects (Pretty *et al.*, 2003; Paillex *et al.*, 2017).

1.2 Development of the water industry and environmental protection in England and Wales

Today, people in England and Wales benefit from a relatively efficient and effective water and sewerage industry, and the development of the water industry can be traced back to the beginnings of the 19th Century (Hassan, 1985; Foster, 2003; Ofwat, 2006). It was influenced by the Industrial Revolution, expansion of urbanisation and increasing demand due to economic development and population increase (Foster, 2003; Ofwat, 2006). The water industry in England and Wales was originally fragmented, with a variety of bodies responsible for different parts of the water cycle (Black and Morrison, 1979; Sheail, 2000; Skelton, 2017).

The United Kingdom has a long history of legislation designed to regulate rivers and their associated fisheries, with earliest records found in the Magna Carta in 1215, for the removal of weirs from the Thames and Medway to benefit both fisheries and navigation (Ayton, 1998). As a result of the Industrial Revolution, fish stocks in many rivers appear to have been in decline (Chapter 2). However, no parliamentary action was taken until a Royal Commission was established in 1860, with a remit to enquire into the Salmon Fisheries of England and Wales (Ayton, 1998). Followed by substantial national investigation into the salmon fisheries condition in many rivers, the Royal Commission Report on Salmon contributed to the passing of the Salmon Fisheries Act 1861 (Ayton, 1998; Skelton, 2017). The Act repealed 33 previous Acts of Parliament, and sought to bring all the legislation together in one Act. It addressed some of the issues such as river obstructions, pollution, illegal fishing and the defective regulation of close seasons (Ayton, 1998).

Some deficiencies in the 1861 Act were remedied with the passing of the Salmon Fishery Act 1865. It allowed Boards of Conservators to be set up, with powers to manage rivers or river systems as defined by the Secretary of State. By 1894, 53 Boards had been set up, covering three quarters of England and Wales. In 1900, Boards of Conservators had been alternatively reconstituted as Fishery Boards (Ayton, 1998). In 1923, the Salmon and Freshwater Fisheries Act 1923 again repealed all previous fishery legislation, and sought to gather it all together in one Act (Meek, 1923). Due to amalgamations and lack of funds to continue to operate in some Boards, only 45 Boards remained by 1948 (Ayton, 1998).

In the early 20th Century there were nearly 200 water providers with a statutory duty to provide piped water to domestic and other consumers. These water providers were either statutory companies, local authorities or joint boards of local authorities (Black and Morrison, 1979). In 1945, there were more than 1,000 local authorities or organizations responsible for water supply and more than 1,300 authorities responsible for sewerage and sewage disposal (Black and Morrison, 1979; Ofwat, 2006). This resulted in poorly integrated and managed water infrastructure with wide variability in drinking water delivery and sewage removal and management (Sheail, 2000; Ofwat, 2006). River Boards were established by the River Boards Act 1948 (Ayton, 1998). The responsibility of 45 Fishery Boards were transferred to 32 River Boards and two conservancies, the Thames and Lee (Howarth, 1987; Ayton, 1998), covering the whole of England and Wales, with responsibilities for land drainage, fisheries and river pollution control previously provided by the local authorities. The act give River Boards powers to administer pollution control across an area defined by watersheds or catchment boundaries (Ofwat, 2006), although the degree to which this was achieved was limited by the need not to severely impinge upon economic activity.

Following growth in the national economy in the 1950s, the demand for water by industry, electricity generation, irrigation and domestic consumption increased substantially. In order to protect water resources and enhance the coordination of different departments, the Water Resources Act 1963 was passed (Ofwat, 2006). The act aims to provide adequate water resources management and to ensure conservation of future water resources (Ofwat, 2006). Two new types of authorities, River Authorities and the Water Resources Board were established to facilitate the aims above (Black and Morrison, 1979).

Under the 1963 Act, the previous 32 river boards were replaced by 27 River Authorities (Ayton, 1998; Ofwat, 2006). Each River Authority took over the powers of the existing river boards, and was responsible for monitoring water quality and protecting water resources in a river basin or a series of river basins (Ofwat, 2006). The Water Resources Board was a national agency, whose aims were to provide advice to government and to the River Authorities on water resources conservation on a national scale (Ofwat, 2006). Despite the role of the River Authorities, they had limited capability to prohibit water pollution. Although they progressively instituted more widespread treatment of sewage, over wider networks, the extent of investment remained limited by the degree of national investment.

In the early 1970s, the government proposed a plan to manage water resources on an integrated river basin basis. A total of ten regional water authorities based on the ten major river basins in England and Wales was established under the Water Act 1973 (Figure 1.3; Ofwat, 2006). The 27 river authorities were abolished and their powers and duties passing to 10 regional water authorities. The regional water authorities were directly responsible for all kinds of water resource management, fisheries, water quality management, pollution prevention, aquatic ecology management, sewerage and sewage disposal (Black and Morrison, 1979; Johnson, 1988; Ofwat, 2006). The regional water authorities were also responsible for water supply in large areas, although a series of local water supply companies remained. For example, the South Essex Waterworks Company (formed 1861) and Southend Waterworks Company (formed 1865) merged to form the Essex Water Company in 1970, which remained independent until it combined with the Suffolk Water Company to form Essex and Suffolk Water Company in 1994. Similarly, a series of local, independent drainage boards also remained, particularly in parts of eastern England.

The regional water authorities continued to make progress in water environment improvement through public investment in sewage treatment, managing industrial and commercial water treatment, and managing water abstraction (Black and Morrison, 1979; Johnson, 1988). Increased regulation of discharge consenting occurred and prosecution of those polluting watercourses became increasingly common. The ability to manage and reduce environmental stressors (especially pollution) acting across whole catchments, from the upper reaches to the estuary and coastal environment, enabled significant recovery for some of the most polluted (especially from industrial and urban sources) rivers such as the Thames, Trent, Mersey and Tyne (Black and Morrison, 1979; Johnson, 1988). However, the combination of environmental regulator and sewage treatment responsibilities within the same organisation also caused a conflict of interest, since some of the most frequent and severe pollution incidents were caused by the water authorities' sewage treatment works (Black and Morrison, 1979). Increasingly this, as well as the limits on investment in water supply and treatment infrastructure through competition for state funds, drove the need for further change in the national water industry and water environment protection.

Figure 1.3 The controlled area of 10 regional water authorities, in England and Wales, 1973-1989. Source from Ofwat (2006).

The regional water authorities lasted until 1989, when the Water Act 1989 provided the mechanism for privatisation of the water supply and sewerage industry (Cowan, 1993). This facilitated large scale private investment to update the ageing infrastructure much of which, in large urban areas, dated from the Victorian period. The water supply, sewerage and sewage disposal parts of the ten English and Welsh regional water authorities became privatised companies (Ofwat, 2006). The regulatory and environmental protection duties remained with the newly created authority, the National Rivers Authority (NRA). The functions of the NRA included monitoring inland and coastal water quality, water pollution control, water resources control and management, salmon and freshwater fisheries, flood defence, recreation, navigation, conservation and harbour authority activities (Ofwat, 2006).

The Environment Act 1995 integrated the functions of the NRA, Her Majesty's Inspectorate of Pollution (HMIP), the Waste Regulation Authorities and certain elements of the Department of the Environment to a new body, the Environment Agency (Slater and Jones, 1999; Ofwat, 2006). The Environment Agency (EA) provides an integrated approach in protecting the environment by combining the regulation of activities affecting land, air and water. For water, the EA is responsible for (i) conservation and pollution prevention in rivers, estuaries and coastal waters, (ii) conserving, redistributing, augmenting and securing proper use of water resources, (iii) flood defence supervision, (iv) maintaining and improving fisheries and (v) promoting the conservation of freshwater and coastal waters, as well as their recreational usage (Slater and Jones, 1999; Ofwat, 2006).

Since 1989, many of the smaller private water companies have merged with the regional water companies. For example, the Essex and Suffolk Water Company referred to above became part of Northumbrian Water in 2000, despite the geographical separation. Although quite strictly regulated, over 70% of England’s water industry, including Northumbrian Water, is now owned by overseas interests. The EA continues to regulate English water company activities in concert with the Water Services Regulation Authority (Ofwat) which controls pricing and investment in the water companies. Increasingly, over recent decades, the EA has taken a more holistic approach to environmental protection of catchments by managing diffuse pollution and climate change impacts, through activities such as land management advice and groundwater monitoring (Ellis and Mitchell, 2006; Wilby and Harris, 2006; Owen *et al.*, 2012). This reflects the historical transition from the greatest impacts on watercourses in the early 20th Century being point source pollution, especially in urban environments, to one where these now include diffuse pollution, climate change impacts, hydrological connectivity, and increasing consumer demand for water (Ellis and Mitchell, 2006; Wilby and Harris, 2006; Owen *et al.*, 2012). Most recently, as part of the devolution process in Wales, EA Wales was merged with the Countryside Council for Wales and the Forestry Commission Wales in 2013 to form Natural Resources Wales, responsible to the Welsh government.

1.3 Water Framework Directive

The Water Framework Directive (WFD), a European Union legal instrument, which became effective on 22 December 2000, aimed to achieve good water status in EU member states for inland surface waters, transitional waters, coastal waters and groundwater by 2015 (European Commission, 2003, 2007a). The Directive's key purposes include: a) protection and enhancement of the status of water resources; b) promotion of sustainable water use based on long-term protection of water resources; c) ensuring the progressive reduction of pollution of groundwater and preventing its further pollution; d) contributing to mitigating the effects of floods and droughts; e) enhancement, protection and improvement of the aquatic environment (European Commission, 2003).

Unlike earlier freshwater condition improvement schemes, WFD emphasized 'good ecological status (GES) or potential', rather than physical or chemical criteria, as the legal basis by which grading would be determined. Like other EU states, the UK did not meet the deadline of achieving good water status by 2015 (UK Technical Advisory Group, 2008a) and the schedule for meeting GES has been extended to 2027. The EA predicted 34% of UK surface water bodies would meet/exceed GES by 2015 (Environment Agency, 2009a). Ultimately, however, just 17% of surface water bodies in England were in the good/high ecological status in 2015 (Table 1.1) (Environment Agency, 2015).The main reasons for not achieving GES include physical modifications to rivers, diffuse source pollution from agriculture and point source pollution from waste water treatment (Environment Agency, 2009a). The Environment Agency believes it is not possible to achieve GES in all UK water bodies by 2027 under current technologies, so it aims to achieve good status in at least 60% of waters by 2021 and in as many waters as possible by 2027 (Environment Agency, 2009a).

Table 1.1 Ecological and chemical 2015 classification for surface waters in England (Environment Agency, 2015). Note the disparity between the proportions of waterbodies meeting good chemical vs good ecological status.

	Ecological status or potential					**Chemical status**	
No. of water bodies	Bad	Poor	Mod	Good	High	Fail	Good
4,679	136	765	2,966	805	7	137	4,542

For WFD, surface water quality status is measured as metrics of ecological, hydromorphological and chemical quality in defined localities known as 'water bodies' (which, for rivers, are georeferenced sections of stream or river [usually ~5-30 km in length] defined for WFD use). The overall ecological grading is determined by the lowest component of these (European Commission, 2003; UK Technical Advisory Group, 2008a); thus waters of good chemical status but less than good ecological or hydromorphological status cannot reach a 'good' class overall. The WFD requires standardised methods to quantify the ecological status of water bodies across five classes of ecological status from bad to high (European Commission, 2003, 2007a; UK Technical Advisory Group, 2008a). These classes reflect different degrees of impact on aquatic species. By using biological indicators, ecological status can be determined and expressed (European Commission, 2007b). After biological quality elements monitoring, the observed parameter values are expressed as an Ecological Quality Ratio (EQR, 0-1, low-high), compared to reference values (European Commission, 2007b).

The biological elements for ecological status classification are composed of phytoplankton, aquatic flora, benthic invertebrate fauna and fish fauna (Birk *et al.*, 2012). In the UK, for fish, the 23 most prevalent native species are used as indicators for assessment (UK Technical Advisory Group, 2008b). These listed fish are classified as of low, moderate and high tolerance to environmental disturbance. After sampling by electro-fishing (normal on streams/rivers) or seine netting methods, fish species counts in running waters are obtained in a single removal for a known area (UK Technical Advisory Group, 2008b). For calculating the EQR, these observed values are compared to reference values, which are determined by modelling and expert judgement (UK Technical Advisory Group, 2008b). In the UK, although these fish community data are not fully quantitative, they provide standardised temporal comparisons for fixed sites at WFD water bodies and can be compared to predictions by models incorporating factors such as distance from source, altitude, channel width and alkalinity. One problem with this approach in England, and throughout much of Europe is that 'reference' conditions are difficult to obtain as most rivers and their biota, especially fish, have been modified by human activities in recent centuries.

1.4 Anthropogenic impacts on fish

Fish populations are threatened by increased anthropogenic activities all over the world. Instream barriers disrupting river connectivity, habitat deterioration and degradation damaging fish production, overfishing and impacts of climate change all pose threats to fish populations (Duncan and Lockwood, 2001; Fenkes *et al.*, 2016; Belliard *et al.*, 2018; Brink *et al.*, 2018). Urbanisation and industrialisation impacts fish communities, especially migratory fish, from many aspects including hydrosystem infrastructure construction, removal of riparian vegetation, flow diversion, irrigation, chemical pollution and sedimentation issues (De Groot, 1992; Fenkes *et al.*, 2016; Brink *et al.*, 2018).

1.4.1 Impacts of pollution

Chemical pollution covers a wide range of impacts and effects on aquatic habitats and biota, including fish. This includes the impact on individual fish and fish populations as they respond to chemical changes to which they are not adapted (e.g. induced ecotoxicological responses) (Saunders and Sprague, 1967; Scott, 2001; Jüergens, 2015; Brink *et al.*, 2018). Historically in urbanised and industrialised areas of Europe, point source pollution was the dominant type of water pollution; it is also the type of water pollution that has been most effectively dealt with across Europe (De Groot, 1992; van Dijk *et al.*, 1994), including in the post-industrial rivers of NE England, such as the Rivers Wear and Tyne (Neal *et al.*, 2000; Kelly, 2002; Baker *et al.*, 2003; Shepherd *et al.*, 2009).

The greatest component of point source urban pollution is organic pollution, mostly resulting from sewage waste. When unoxidized sewage waste enters the environment (typically to rivers/estuaries), it causes dramatic oxygen depletion, proportional to its biochemical oxygen demand (BOD) for decomposition (Kumar and Reddy, 2009). Such waste may also contain high levels of unoxidized nitrogen as ammonia, which has a high toxicity to aquatic organisms and, inevitably, high levels of nutrients including phosphorus, since it is largely the product of human urine and faeces (Jarvie *et al.*, 1998; Neal *et al.*, 2005). Furthermore, sewers also collect organic and inorganic material (e.g. tyre rubber, heavy metals) from land and road runoff and permitted chemical outputs from manufacturing and light industry (Neal *et al.*, 2005). Therefore, sewage waste and combined sewer overflow (CSO) outputs contain a wide variety of biological pollutants and contaminants.

Sewage treatment processes reduce the levels of pollutant and contaminant materials reaching the environment, such that the sewage outfall water must fall within permitted amounts, so that diluting effects of the river, estuary or marine environment adequately mitigate the remaining pollutant impacts. The extent of such treatment (primary, secondary, tertiary) depends on relative costs, population equivalent (PE) and the degree of environmental standards (i.e. tertiary treatment is required in sensitive areas, for example phosphate stripping in nutrient-sensitive areas) applied (Defra, 2002, 2012). In the UK, there are around 9,000 waste water treatment plants linked to the largest collection systems, and approximately 1,900 of these plants serve agglomerations of greater than 2,000 PE to freshwaters and estuaries, and greater than 10,000 PE made to coastal waters (Defra, 2002, 2012).

Increasingly, agricultural pollution, especially through diffuse sources, is replacing industrial and domestic pollution as the greatest cause of poor water quality in many rivers. Agriculture is one of the most important components in the global economy and it is also the largest freshwater user, involving 70% of surface water supplies on the global scale (Ongley, 1996). Today, largely as a result of the intensification of agricultural practices, diffuse water pollution from agriculture is one of the major contributors to surface and ground water pollution in the UK. Agriculture uses 70% of the UK's land area (Defra, 2017). It is estimated that agricultural pollution has contributed approximately 70% of the nitrates, 28% of the phosphates and 76% of fine sediments to UK rivers (Edwards and Withers, 2008; Collins *et al.*, 2009). This has led to the situation where in England, although some previously industrially polluted rivers such as the Tyne and Mersey have seen dramatic improvements in water quality, many rural rivers previously of high quality, for example the River Wensum (Norfolk) and River Axe (Dorset, Somerset, Devon), have deteriorated due largely to diffuse pollution (Natural England, 2015; Cooper *et al.*, 2020).

Pollutants from agriculture are commonly classified into four types: fertilisers, pesticides, sediments and faecal bacteria (Ongley, 1996). There are high levels of nitrogen, potassium and phosphorus in agricultural fertilisers (Environment Agency, 2007a), whether these are from mineral sources (granular fertiliser) or livestock waste (slurry spreading). Excess nutrients, especially soluble phosphorus, entering a river system can lead to increased, sometimes explosive, growth of plants and algae, known as eutrophication (Wortmann *et al.*, 2005; Environment Agency, 2007a). Excessive growth of macrophytes and algae can result in poor water quality, including low dissolved oxygen at night, resulting in fish deaths (Defra, 2019), but also has a more pervasive, chronic effect by causing species replacement, community and ecosystem modification. For example, in salmonid-dominated rivers, algal growths can cover the stream bed altering its physical structure for benthic organisms, but also limiting water flow through and dissolved gas exchange with gravel/cobble beds. Decomposing algae increase oxygen demand at the stream bed and contribute to hypoxia within the hyporheic zone (Mallin *et al.*, 2006; Cox and Whitehead, 2009), contributing to mortality of salmonid eggs and alevins (yolk-sac fry) (Kemp *et al.*, 2011).

Increased levels of fine sediments from agricultural runoff can blanket and infiltrate gravel beds, smothering salmonid spawning grounds, decreasing hyporheic flow and oxygen transport to eggs, and decreasing the productivity of salmonid rearing habitat (Tappel and Bjornn, 1983; Reiser and White, 1988; Armstrong *et al.*, 2003). Manure spreading, livestock slurry and silage effluent are also pollution sources in agriculture (see above, regarding eutrophication). These activities often result in high levels of contamination of receiving waters by pathogens, metals, phosphorus and nitrogen, and potential contamination of ground water (Ongley, 1996; Wortmann *et al.*, 2005).

Pesticides can be washed into the river system through rainfall or get into the surface water directly due to pesticide spraying. Pesticides can cause a series of negative impacts to the aquatic system such as surface water contamination; death of fish and invertebrates; tissue damage and cancer; reproductive failure; physical deformities (Ongley, 1996; Liess and Schulz, 1999; Geeraerts and Belpaire, 2010). Chemically contaminated sediments (e.g. heavy metals, pesticides, industrial organochlorines such as polychlorinated biphenyls [PCBs] can accumulate on the bottom of the stream and reduce biodiversity in river beds through sublethal or lethal toxic effects (Geeraerts and Belpaire, 2010; Rose *et al.*, 2015).

1.4.2 Impacts of habitat modification

Rivers and their riparian zones play a key role in the maintenance of aquatic biodiversity (Dynesius and Nilsson, 1994). UK rivers have suffered negative anthropogenic impacts for centuries (Figure 1.2). Physical modifications affect 39% of WFD water bodies in England (Environment Agency, 2015). These modifications such as river straightening, gravel extraction, dredging of sediment, flood defences, land drainage and irrigation systems, dams and weirs alter the shape and size of rivers and their riparian zone, increase river degradation and cause habitat fragmentation (Dynesius and Nilsson, 1994; Environment Agency, 2015). The main processes impeded, resulting in the modifications described above, are loss of hydrological, sediment, nutrient and ecological connectivity. Habitat modification and fragmentation have been recognized to be major threats to aquatic species abundance and biodiversity (Larinier, 2001; Malmqvist and Rundle, 2002; Nilsson *et al.*, 2005; Dudgeon *et al.*, 2006; Noonan *et al.*, 2012). Habitat modification is a fundamental reason for losses of many fish species and reductions in fish population size along many rivers, especially for fishes with specialist needs, such as rheophilic species (i.e. those living in fast-moving waters), and lithophilic species (i.e. those requiring stony reproduction habitat).

Channelization includes river straightening, diversion, deepening and the creation of artificial channels. In England and Wales, river channelization was undertaken increasingly from the Middle Ages onwards to mitigate flood issues, improve agricultural drainage, improve navigation and prevent bank erosion (Brookes *et al.*, 1983; Millidine *et al.*, 2012). These activities reached a peak in the mid-20th Century, when it was estimated that a total of 8,504 km of river network were channelized in England and Wales (Brookes *et al.*, 1983). Channelized rivers normally have uniform cross-section shapes and bottom substrates dominated by fine sediments that lead to less varied in-stream habitats for different aquatic biota including invertebrate and fish (Negishi *et al.*, 2002; Harrison *et al.*, 2004; Millidine *et al.*, 2012). In addition, channelized rivers often require regular dredging to control sediment deposition. But this action may lead to unintended bank erosion and other negative effects on aquatic ecosystems.

Removal of coarse substrates such as pebbles and gravels can damage the spawning grounds of species such as salmon (*Salmo salar*), brown trout (*Salmo trutta*), bullhead (*Cottus gobio* species complex, including *C. perifretum*) and river lamprey (*Lampetra fluviatilis*) (Figure 1.4). Apart from fish, dredging also poses threats to endangered UK species such as the freshwater pearl mussel (*Margaritifera margaritifera*) (Cosgrove and Hastie, 2001) and the white-clawed crayfish (*Austropotamobius pallipes*) (Demers and Reynolds, 2003). It was reported that a dredging operation destroyed the entire freshwater pearl mussel population in a north-west Wales river (Killeen *et al.*, 1998; Cosgrove and Hastie, 2001). Similarly, gravel extraction also causes damage to rivers by altering river topography and sediment composition (see Chapter 2 for details).

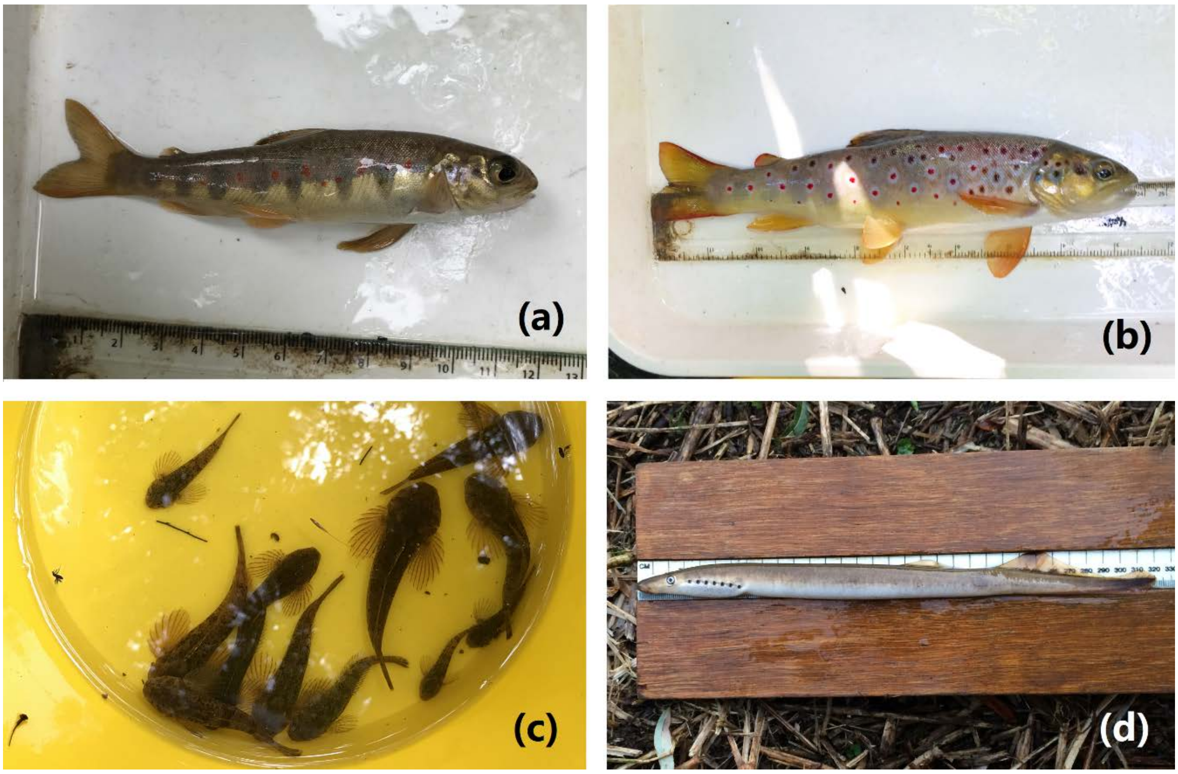


Figure 1.4 Examples of gravel-cobble spawning fishes, (a) Atlantic salmon [parr life cycle stage shown], (b) brown trout, (c) bullhead, (d) river lamprey.

Instream wood and vegetation removal is another human activity which has received increased attention in recent years (Old *et al.*, 2014). Instream macrophyte vegetation is especially important in lowland rivers such as the chalk rivers of southern England (Flynn *et al.*, 2002; Cotton *et al.*, 2006), less so in the Pennine rivers of northern England so the emphasis that follows is on woody material, originating from the riparian zone. Instream wood, also known as large woody debris (LWD) is pieces of dead wood larger than 0.1 m in diameter and 1.0 m length, and can refer to entire trees, branches or root plates that have fallen into rivers (Linstead and Gurnell, 1998). Instream wood is one of the vital elements for functioning river ecosystems and is considered as important as riparian/instream vegetation and sediment (Nakamura *et al.*, 2017; Ruiz-Villanueva and Stoffel, 2017). However, instream wood has been extensively removed from river systems in some countries in order to improve navigation and flood control (Wohl, 2014; Elosegi *et al.*, 2017). Instream wood removal increases channel conveyance and reduces flow resistance within the channel, resulting in the reduction of secondary channels, floodplain area and bank stability (Wohl, 2014). Rivers with less instream wood tend to become wider, straighter with less habitat diversity and reduced lateral connectivity (Wohl, 2014).

1.4.3 Impacts of anthropogenic in-stream structures

Longitudinal barriers cause issues for both upstream and downstream fish migrations and dispersal (Lucas and Baras, 2001). Although some fish species are migratory (Lucas and Baras, 2001) all river organisms rely on dispersal for recolonization, metapopulation maintenance and gene flow (Radinger and Wolter, 2014; Tummers *et al.*, 2016; Wilkes *et al.*, 2019). Barrages, flood-control dams, tidal barrages and sluices, pumping- and hydropower stations are all examples of potential barriers to fish movement (Brink *et al.*, 2018). Structures such as weirs and dams are the main reason for river fragmentation; these reduce the availability of key habitats for fish and other aquatic species, and obstruct migration and dispersal of aquatic biota including fish. Single, small-scale barriers like low-head weirs (0.5 – 4.0 m) usually have lesser impacts on fish populations compared to large dams, but because there are many more small obstacles (Jones *et al.*, 2019), their cumulative impacts may be much greater (Lucas *et al.*, 2009). It has been estimated that there are nearly 25,000 weirs and similar structures in the rivers of England and Wales (Elbourne *et al.*, 2013), although this is an underestimate (Jones *et al*. 2019).

Globally, many fish species have been impacted by in-stream barriers and these have caused substantial declines in their populations. The species most strongly influenced by engineered cross-channel structures are diadromous species (migrating between freshwater and the sea e.g. Atlantic salmon) and potamodromous species (migrating solely within freshwater e.g. barbel *Barbus barbus*), because their migrations can be prevented, restricted or delayed by cross-stream obstacles. Small artificial barriers like weirs with a head of just a few tens of centimetres can also contribute to the decrease of some small migratory cyprinid species and non-migratory sculpins such as European bullhead (*Cottus gobio*), because those fish do not display leaping or strong swimming abilities (Peter, 1998; Lucas and Baras, 2001). In North-Western Europe, Atlantic salmon have been extirpated from at least 300 rivers, and Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) has been extirpated from most European rivers due to the obstruction of their migratory paths and degradation and alteration of rivers (De Groot, 1992; Holčik, 1996;

Raat, 2001; Brink *et al.*, 2018). In Sweden, stocks of brown trout have declined by two thirds in Lake Vänern due to migration obstructions (Brink *et al.*, 2018).

1.5 River restoration

With the increasing threats to rivers and their biota, there is an increasing need for river ecosystem management. The term river restoration, increasingly widely used in the last 20 years, has become associated mainly with actions to restore physical habitat and channel form to nearer natural conditions (Addy *et al.*, 2016). Multiple measures can be implemented separately or combined with each other for restoring river habitat and biodiversity. The aims of river restoration are to repair and restore the natural physical process, river features, physical habitats, then assist the degraded river system to function again (Addy *et al.*, 2016). If restoration is successful, the river system should have the adaptive ability to maintain its restored condition (Speed *et al.*, 2016).

Yet Figure 1.1 illustrates that deterioration of the ecological function of rivers is associated with chemical as well as hydromorphological impacts. Thus, effective restoration of ecological function in rivers necessitates water quality improvements, through pollution control, as well as by hydromorphological rehabilitation. The return of large populations of migratory salmonids (with low tolerance to poor water quality) has been achieved in post-industrial English rivers only as a result of massive investment to improve the quality of waste water releases in these rivers. With the decline of heavy industry, reduced effluent, and improvement in water quality, especially in the lower river reaches and estuary, Atlantic salmon have recolonized (Perrier *et al.*, 2010; Ikediashi *et al.*, 2012), but spawning success and populations remain low in many potential nursery areas due to migration barriers, degraded habitat and/or poor water quality. Similar issues affect other fish species to a somewhat lesser but nevertheless significant degree (Tummers *et al.*, 2016). Domestic and industrial effluent water quality improvements are mainly managed through the water industry companies in conjunction with environmental regulators, but restoration of hydromorphological functionality has been a more recent objective, in the EU stimulated especially by WFD targets (Haase *et al.*, 2013; Schmutz *et al.*, 2016; Rinaldi *et al.*, 2017).

1.5.1 Connectivity restoration in rivers

Natural ecological functioning of many rivers in the UK has been damaged by multiple issues for several centuries. The ecological problems became serious due to the rapid development of industry and urban environments. Obstacles like dams and weirs blocked the migration paths for different fish species, resulting in increased river fragmentation. The development of agriculture also resulted in the loss and alteration of riparian zone and floodplain, as well as reducing the refuges for riverine animals and damaging the habitats for fish, invertebrates, aquatic mammals and a variety of other species. To restore river longitudinal connectivity, one of the common measures is to recreate fish passes or biota passes (Silva *et al.*, 2018), although this does not solve the origin of the problem. Increasingly solutions are sought that as well as passing fish and other biota, also reinstate natural transport processes within the river, and support natural habitats, typically through barrier removal (Kemp and O'Hanley, 2010; O'Hanley, 2011; King and O'Hanley, 2016). However, lateral connectivity restoration is also needed, especially in floodplain regions since ecological processes here are intrinsically linked with lateral hydrological connectivity and wetland habitats (Schiemer *et al.*, 1999; Bolland *et al.*, 2012). The WFD requires the provision of free migration of fish in river systems, stimulating increased efforts to provide longitudinal and lateral connectivity for fish between habitats.

With regard to longitudinal connectivity, which is the biggest connectivity problem for migratory fishes, and in upland rivers with rheophilic fish communities (those typical of and adapted to rapid, flowing water), the potential effect of each in-stream obstacle needs to be recorded so that problematic sites can be selected for carrying out connectivity restoration projects (Ovidio *et al.*, 2007). Based on connectivity restoration research, barrier removal may be the most efficient way to increase fish populations compared with other methods (Kemp and O'Hanley, 2010; Birnie-Gauvin et al., 2017c), restoring longitudinal connectivity and reinstating more natural habitats in terms of depth, flow and substrate variation (Elbourne *et al.*, 2013). Barrier removal is considered to be the most cost-effective and sustainable method in river habitat restoration, because it can restore river morphology, sediment, organic matter, aquatic biota and provision of upstream and downstream movement (Larinier, 2001; Elbourne *et al.*, 2013). From a fish passage perspective, the barrier removal allows fish species to recolonize depopulated upper reaches of a river and contribute to the broader restoration of riverine processes (Brink *et al.*, 2018). Apart from targeted species, barrier removal also benefits other non-target species and weaker swimming fish species which may struggle to ascend fish passes and other restorative measures (Hitt *et al.*, 2012; Hogg *et al.*, 2015; Kornis *et al.*, 2015; Brink *et al.*, 2018). However, if a weir/structure cannot be removed for some reasons (e.g. economic, historical, societal or political restrictions), mitigations such as fish passes need to be considered.

Fish passes (= fishways) can be installed at river barriers to support upstream and/or downstream migration (Noonan *et al.*, 2012), various designs of which are available. The working principle of an upstream fish pass is to dissipate the energy of flowing water by walls, baffles or vanes (or natural structures such as rocks), so the fish can ascend past the structure much more easily (Clay, 1995). The upstream fish pass, designed for certain anadromous species (e.g. salmonids), is well developed (Larinier, 2001), but their efficiency for other species such as potamodromous (e.g. barbel) or catadromous species (e.g. European eel *Anguilla anguilla*) is poorly understood. Downstream fish passes also need to be developed, because for small obstacles it is often assumed that fish will pass over them unimpeded but this is often not the case (Gauld *et al.*, 2013). Calles and Greenberg (2009) emphasize that longitudinal connectivity for migratory river fish, especially diadromous ones, needs to be functionally two-way (upstream and downstream), despite the overemphasis on upstream fishways. A successful ecological restoration project on a river should not only achieve effective connectivity for one or a few species, but also for a wide range of species (Lucas and Baras, 2001; Tummers *et al.*, 2016). However, it may be necessary to design fish pass or connectivity restoration solutions to limit passage and colonisation by non-native invasive species, such as signal crayfish (*Pacifastacus leniusculus*) (McLaughlin *et al.*, 2013; Rosewarne *et al.*, 2013).

Apart from a highly engineered ‘fish pass’, an easement is a pragmatic solution to fish passage which generally falls outside the formal fish pass authorisation process (Armstrong *et al.*, 2010). These are usually relatively small structures, on streams rather than large rivers. The term “easement” is not widely recognised internationally. It is almost exclusively a UK term, developed by the Environment Agency to distinguish formal fish passes (that require approval in design and installation by a formal committee – hence expensive, and slow to process) from semi-formal modifications (easements), often nature-like in form, that do not require formal committee approval, so are much quicker and cheaper to complete. In Britain, easements are usually built by non-governmental, not-for-profit organisations (e.g. Rivers Trusts) in order to reduce the obstructing effects of the many small obstacles occurring in British rivers.

1.5.2 Habitat restoration in rivers

Habitat degradation is one of the biggest problems in protecting the biodiversity and ecosystem functioning of river systems. Some anthropogenic river modifications such as channel straightening, relocation and dredging often cause habitat alteration, degradation and loss (Speed *et al.*, 2016). In order to mitigate those negative impacts and return the river habitat towards a more normal condition for aquatic species, a wide variety of efforts in river habitat restoration have been carried out all over the world (Jansson *et al.*, 2007; Nilsson *et al.*, 2015, 2016). For achieving successful habitat restoration, both riparian zones and in-stream habitat need to be considered. Barrier removal helps to restore habitats in rivers, especially low-gradient ones, because it removes artificially pooled areas and facilitates natural sediment transport (Birnie-Gauvin *et al.*, 2017a). Beyond this, other widely used river habitat restoration methods include re-introduction of large woody debris, river bed control structures, revegetation of the riparian zone and geomorphological reconstruction (Floyd *et al.*, 2009; Speed *et al.*, 2016; Bašić *et al.*, 2017). Consideration of several of these categories is given below, as these are the cheaper and more frequently adopted river restoration methods, while geomorphological construction requires large-scale mechanical actions such as channel reworking and regrading.

1.5.2.1 Large woody debris management

Re-introduction of LWD is considered to be a major method in river habitat restoration, and it has been increasingly used in recent years, particularly for the benefit of salmonid fish species (Cederholm *et al.*, 1997; Solazzi *et al.*, 2000; Elosegi *et al.*, 2017). The LWD installation method is both environmentally friendly and inexpensive, it can accelerate river habitat rehabilitation and also have various benefits to lotic systems. LWD can stabilise the river bank, reduce river bed erosion and trap sediments, gravels and silts, raise bed levels, and help create pools, riffles and backwater sequences (Swanson and Lienkaemper, 1978; Bisson *et al.*, 1987; Gurnell *et al.*, 2005). For example, log jams can increase the biological productivity in rivers and form complex river morphology in the form of pools, gravel bars and covers (Bisson *et al.*, 1987; Montgomery and Buffington, 1997; Elosegi *et al.*, 2017). LWD can also remove fine silt from the river system by creating silt 'benches' immediately upstream, and this process helps to maintain fast flows in the river and can prevent gravels, which provide suitable spawning grounds for fish like salmon and trout, being covered by silts. The loss of LWD can result in declines in fish abundance, average fish size, and standing biomass of salmonid fish species (Coulston and Maughan, 1983; Dolloff, 1986; Fausch and Northcote, 1992).

Large Woody Debris can provide shelters and refuges for salmonids and other fish to hide in (Floyd *et al.*, 2009). They can also create feeding, spawning grounds and nursery sites such as gravel beds, pools and undercut banks for fish (Coulston and Maughan, 1983; Cederholm *et al.*, 1997; Solazzi *et al.*, 2000; Floyd *et al.*, 2009; Elosegi *et al.*, 2017). LWD offers a wide range of spaces for algae, microbes and invertebrates to colonise, which increases the aquatic biodiversity and supplies plenty of food for other aquatic creatures (Cederholm *et al.*, 1997). Based on studies in the Elwha River in Washington State, USA, engineered log jams are useful in increasing juvenile salmon densities for restoring juvenile salmon habitat (Pess *et al.*, 2012). For streams with less woody debris, introduction of LWD was considered to be an effective method in restoring salmonid populations (Floyd *et al.*, 2009).

1.5.2.2 Gravel management

For gravel-cobble spawning fishes such as salmonids, lampreys, cottids (Figure 1.4) and some cyprinids, a high quality spawning and rearing habitat usually consists of a mixture of sand, gravel, cobbles and boulders (the exact mixture varying according to species and life stage) and it is fundamental to the productivity of salmonids (Rosenau and Angelo, 2000; Armstrong *et al.*, 2003). However, human activities such as mining, damming, sand and gravel extraction, dredging and dyking have degraded lots of spawning grounds for fish in streams (Kondolf, 1995; Rosenau and Angelo, 2000; Larinier, 2001). For example, gravel extraction from riverbeds changes the riverbed morphology, depletes instream gravel, degrades the aquatic habitat, causes erosion of gravel bars downstream and destroys riparian vegetation (Collins and Dunne, 1989; Kondolf, 1997). In order to rehabilitate the natural sediments and mitigate the negative impacts on streams, gravel management actions such as gravel additions and gravel jetting have been used (Wheaton *et al.*, 2004; Twine, 2013; Mueller *et al.*, 2014; Bašić *et al.*, 2017).

Gravel addition is a method to add a particular size of gravel into a stream to replace the missing sediments (Mueller *et al.*, 2014). On the Rhine River, 170,000 tonnes of gravel are artificially added to the river below the Barrage Iffezheim annually in order to prevent incision (Kondolf, 1995). On the Upper Sacramento River below Keswick Dam, artificial gravel was added to the river for salmonids to spawn on from 1988 to 2000 (Kondolf, 1995). These projects can provide short-term habitat for salmon, but the gravels placed below the dam have washed out during high flows, so it requires continued addition of more imported gravel (Kondolf, 1997). In this regard it treats the symptom, rather than the cause and so has limited efficacy (Mueller *et al.*, 2014).

If the spaces between streambed gravel have been clogged by fine sediment, salmonid eggs and newly hatched alevins may suffocate due to low oxygen content and juvenile salmonids lose a key food source and shelter (Tappel and Bjornn, 1983; Louhi *et al.*, 2008; Kemp *et al.*, 2011). In order to remove the fine sediment from the gravel beds, high pressure water jetting (= gravel jetting) is sometimes used before the spawning season, although it is a labour intensive method which must be used annually, and silt ingress can still occur during the period of egg incubation (Twine, 2013; Bašić *et al.*, 2017).

1.5.2.3 Restoring riparian vegetation communities

Riparian zones are the interface between land and aquatic environments, processing nutrients, delivering organic matter to a stream, stabilizing soils and providing habitats for both terrestrial and aquatic biota (Roni *et al.*, 2005). However, some human activities like agriculture, industry, grazing, fencing, logging and transportation may cause riparian zone degradation. In order to restore the degraded riparian habitat, riparian silviculture treatments have been widely used to improve the riparian conditions (Roni *et al.*, 2005). These treatments include seeding, planting and also removal of invasive plant species. Restoring the riparian zone can increase stream shading (and hence help protect against climate change impacts of increasing water temperatures), improve water quality, help to rehabilitate wooded wildlife corridors, reinforce the river bank and deliver organic matter into the stream (Addy *et al.*, 2016). Planting native vegetation can rehabilitate the characteristics of riparian vegetation communities, improve self-recovery ability of physical habitat and provide food and nutrients for the fish and invertebrate communities (Addy *et al.*, 2016).

1.5.3 Effectiveness of restoration

Despite many attempts at stream and river restoration, especially on small (tens to hundreds of metres) and medium (several kilometres) scales, good quality empirical data concerning the effectiveness of interventions are sparse relative to the number of schemes (Nilsson *et al.*, 2016; Tummers *et al.*, 2016). This is partly because of limited budgets, partly because budgets are normally directed at the restoration work rather than scientific evaluation, and partly due to the difficulties inherent in measuring the effects of restoration projects relative to processes unrelated to the restoration actions. Increasing numbers of studies are measuring the effectiveness of fish passage and connectivity measures (Noonan *et al.*, 2012; Tummers *et al.*, 2016; Lothian *et al.*, 2020), but the link between passage, dispersal at site level and fish population change at the reach or catchment scale is poorly known, and most such studies are for just a few species. Similarly, evidence concerning the benefits of river habitat improvement measures for fish and other biota are few (van Zyll De Jong *et al.*, 1997; Pretty *et al.*, 2003; Floyd *et al.*, 2009; Verdonschot *et al.*, 2016). In both cases, well-controlled experiments over appropriate timescales (natural recovery may take many years) are difficult to achieve. Often, multiple stressors operate and may be changing on different spatial and temporal scales, again making measurement of changes difficult. Nevertheless, in post-industrial rivers, recording longer term changes in biota (such as fish) is necessarily part of the process of determining the response of these to changing conditions, including adaptive management measures intended to continue improvements towards good ecological conditions (Birnie-Gauvin *et al.*, 2017b).

1.6 Aims and objectives of the study

The aims of this study were to evaluate the changes in fish populations in post-industrial rivers, such as those in NE England, and to determine the roles of river restoration projects, particularly through water quality improvement and connectivity restoration, in facilitating ecological improvements to the fish communities of these rivers. This thesis evaluates historical changes and current trends in fish populations in the Rivers Tyne, Wear and Tees and sets them in the context of their decline in water quality and river habitat, and their subsequent recovery. The thesis also incorporates empirical studies on the distribution and numbers of obstructions in the Rivers Wear and Tees; the effect of a barrier removal in a tidal tributary of the Tees; and effects of partial connectivity restoration in several tributaries of the River Wear.

In the context of the aims indicated above, the following objectives were generated:

1) Determine the historical changes in fish communities and fish abundance, particularly of anadromous salmonids, in upland Northeast English post-industrial rivers (Tyne, Wear, Tees), in relation to environmental quality and other relevant factors such as fisheries, in these rivers.

2) Determine the degree to which the barrier inventories of post-industrial rivers are incomplete, potentially hindering river restoration actions as a result. The context of this objective is that in England, the national river barrier inventory used for management and longitudinal connectivity restoration planning was produced by the Environment Agency. However, I predict that, based on Jones et al. (2019) only a small proportion of in-stream barriers were recorded in the national database. I test this by using walkover surveys on stratified sections of the Wear and Tees catchments.

3) Test, using a case study, the degree to which barrier removal may be a powerful tool for river restoration in small tidal creeks and streams. I predict that tidal barrier removal in small streams linked directly to tidal reaches results in the change of habitat from impounded, lentic water to more diverse habitat, with associated rapid changes in the fish community in the formally impounded zone and benefits the recolonisation of diadromous fishes (e.g. European eel) in tidally-linked streams.

4) Test, using case studies, the degree to which subcatchment-scale connectivity restoration (improving stream connectivity and/or fish passage at multiple sites within a subcatchment) can restore rheophilic fish communities towards expected conditions. I test this on several tributaries of the River Wear where barrier removals and fish passage mitigations have been undertaken, by comparison with reference conditions and/or over time.

1.7 Chapter outlines

Following the General Introduction this thesis comprises one literature review and data combined chapter (Chapter 2), three data chapters (Chapters 3, 4 and 5) and a General Discussion (Chapter 6). Chapter 2 addresses Objective 1. This chapter uses historical literature and secondary data to assess the decline and recovery of fish, water quality and aquatic habitat in the Tyne, Wear and Tees catchments. Chapter 2 also provides background information for remaining chapters.

Chapter 3 addresses Objective 2, assessing the level of completeness of the current national barrier inventory. This study was based on a detailed walkover survey on the main River Wear and Tees as well as 20 of their tributaries. Both artificial and natural barriers were recorded during the survey then compared with the national barrier inventory. Outcomes of this study should be helpful for river management agencies in planning future connectivity restoration works in Wear and Tees catchments.

Chapter 4 addresses Objective 3. A before-after study was carried out in a coastal lowland stream in the Tees catchment, NE England. It measured the short-term changes of aquatic habitat, benthic macroinvertebrates and fishes after tidal barrier removal. Outcomes of this study should provide a better understanding of how connectivity and habitat restoration may facilitate ecological improvements to the fish community.

Chapter 5 aims to address Objective 4. This study determined the effects of multiple (but partial) connectivity restoration works on local fish communities in three degraded streams in the Wear catchment, NE England. Outcomes of this study may be helpful for understanding the recovery of migratory salmonids and other fish species after sub-catchment scale connectivity restoration. Finally, Chapter 6 integrates the key findings from Chapters 2 to 5, and considers implications for future river restoration management and research.

Chapter Two

History of the decline and recovery of fishes in post-industrial rivers in Northeast England

Summary

In order to contextualize the condition of post-industrial rivers in North East England for empirical studies in this thesis (Chapters 3-5), a historical review of the decline and partial recovery of the rivers Tyne, Wear and Tees, and their fish stocks, was carried out. The review concentrated on the period through the Industrial Revolution to the current day and combined historical information from published sources with collation and presentation of secondary data sourced from government agencies.

Although heavy metal mining occurred in the upper Tyne catchment for centuries before, the lower Tyne, particularly the estuary, was severely polluted from the 1850s to the 1950s due to urban development and industry around Newcastle upon Tyne and Gateshead, without adequate pollution treatment or control. Both Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) became functionally extinct from the river in the early 20th Century. The abundance of salmon and sea trout in the Tyne increased progressively since the 1960s when the pollution was eased through progressive actions to treat waste water, and through industrial decline. The building of Kielder Reservoir, completed in 1981, without a fishway, obstructed key spawning habitat for migratory salmonids in the North Tyne, theoretically mitigated by hatchery releases of offsprings from Tyne broodstocks. The annual rod catch of salmon and sea trout dramatically increased from the 1980s to the 2010s, but the increase has stabilized in recent years, with a peak in 2010 (sea trout) and 2011 (salmon). Nevertheless, the Tyne is now the most productive salmon river anywhere in England. Knowledge of historical changes to the remainder of the river's fish community is very poor until the last few decades. However, 82/123 (66.7%) of Tyne Water Framework Directive (WFD) water bodies failed to reach good ecological condition in 2019, with the greatest pressures coming from hydromorphological modification and pollution from abandoned mines.

In the River Wear, salmon and sea trout migration was seriously affected by instream barriers in the early 19th Century. The Wear was heavily polluted due to coal and metal-mining activities in the mid-19th Century. Pollution was exacerbated by release of untreated human sewage in many mining towns, often situated close to watercourses.

Closure of coal mines around the Wear occurred throughout the 20th Century, with the last ones, in the east of the catchment, closing in the 1980s but with minewater pumping and treatment of those continuing. Other heavy industry decreased and progressive improvements in sewage and wastewater treatment occurred from the 1960s onward. Similar to the Tyne, an increase in salmon and sea trout abundance occurred in the Wear, resulting in recovery of what had been a functionally extinct salmon population. That increase has levelled off in recent years, and there is some evidence of a recent decline, but the Wear is the second-most productive salmon fishery in England. Knowledge of historical changes to the remainder of the river's fish community is very poor until the last few decades. However, 58/64 (90.6%) of Wear WFD water bodies failed to reach good ecological condition in 2019, with the greatest pressures coming from urban, wastewater and minewater pollution sources and from hydromorphological modification.

The upper Tees was subject to intense heavy metal mining for centuries before the Industrial Revolution, but industrial pollution since the mid-19th Century seems to have been responsible for a major decline in salmon and sea trout. Much of the estuary's wetland habitat was reclaimed for heavy industry (especially steel production and the chemicals industry). Impoundment of upper Tees tributaries was extensive between the late 19th Century and the 1980s. Water treatment in the Tees improved from the 1970s and since 1982 salmon and sea trout rod catches have slowly increased but have not followed the rapid trajectory of the Tyne and Wear. A tidal barrage, 16 km upstream of the river mouth, was opened in 1995. This barrage aimed to limit the upstream movement of grossly polluted water in the lower estuary and facilitate urban redevelopment, but it also inhibits fish migration. Despite some relatively detailed biological surveys in the 1930s, knowledge of historical changes to the remainder of the river's fish community is also very poor until the last few decades. However, 74/87 (85.1%) of Tees WFD water bodies failed to reach good ecological condition in the 2019, with the greatest pressures coming from hydromorphological modification and diffuse pollution sources.

2.1 Introduction

Globally, many rivers have suffered severely from pollution, habitat change and connectivity loss (Chapter 1). This situation has worsened over the last 200 years, and especially in many parts of the world within the last few decades, when human development pressures have been at their greatest (Maitland, 1995; Dudgeon *et al.*, 2006). Intense pressures have changed the ecological condition of rivers and led to the deterioration of river ecosystem functioning, reflected in declines of native biodiversity including fish populations. However, such impacts have not increased progressively in all areas of the world. In Western Europe, for example much of the damage to rivers occurred during the agricultural and Industrial Revolutions when rivers became highly fragmented and large amounts of urban and industrial pollution occurred (Hoffmann, 1996; Downward and Skinner, 2005; Walter and Merritts, 2008; Nützmann *et al.*, 2011; Winiwarter *et al.*, 2016). Examples of some of these declines and, in some cases, ecological recovery, are given below, highlighting that much of the damage that needs to be repaired is the carryover effect of activities from many decades or even centuries ago.

The River Rhine, one of the major European rivers, stretching from Switzerland to the Netherlands, is considered to be one of the most anthropogenically affected river systems in Europe (Lelek and Köhler, 1990; Lenders, 2017). The river has been heavily modified with channelization for more than 100 years and the majority of floodplain area has completely lost its ecological function (Lelek and Köhler, 1990; Raat, 2001). Within the Netherlands region, all branches of the River Rhine are canalized and there is no actual river delta left any more (Raat, 2001). A total of seven native fish species had been extirpated in the Rhine by 1990, including European sturgeon (*Acipenser sturio*) (Lelek and Köhler, 1990). Atlantic salmon (*Salmo salar*) are functionally extinct in the Rhine although attempts are being made to restore them (see below). Salmon showed a decreasing abundance trend since the 1500s, exacerbated by the later effects of increased industrialization (De Groot, 1992, 2002; Lenders, 2017). Salmon captures in the Rhine–Meuse estuary decreased from an average number of 70,000 fish in the 1880s to zero in the 1950s (De Groot, 1992; Raat, 2001). There are several reasons for the decline of the salmon population: construction of in-stream barriers inhibiting migration, decline of spawning and nursery grounds by gravel extraction, domestic and industrial waste water pollution and intensive fisheries (De Groot, 1992; Raat, 2001). Houting (*Coregonus oxyrhynchus*) used to be present in high abundance in this river, but have disappeared since 1940. The decline of the houting population is probably due to loss of spawning and nursery habitat (Raat, 2001). The allis shad (*Alosa alosa*), was found in high numbers in the lower parts of the Rhine and Meuse in the 1920s, but it became extinct from the rivers in the Netherlands region (Raat, 2001). The reason for the extirpation of the shad population is similar to that for salmon.

In the 1960s and 1970s, much of the River Rhine was very polluted and river habitat was on the edge of total deterioration. In addition, the Sandoz chemical spill at Basel, Switzerland in 1986 caused catastrophic effects on the river biota, including fish (Giger, 2009). The incident eliminated the entire eel (*Anguilla anguilla*) population in the 400 km reach downstream of the chemical plant and caused significant damage to other fish populations such as grayling (*Thymallus thymallus*), brown trout (*Salmo trutta*), pike (*Esox lucius*) and zander (*Sander lucioperca*), and to many other organisms (Güttinger and Stumm, 1992; Giger, 2009). The pollution incident also facilitated the colonisation of invasive species through occupation of vacant ecological niches due to the loss of native biota. It has been estimated that more than 90% of biomass in the Rhine is composed of invasive species (Giger, 2009). A series of major restoration projects have attempted to restore the ecological quality of the Rhine, partly stimulated by the Sandoz incident, but also by the Water Framework Directive (WFD). “Salmon 2000”, followed by “Salmon Come Back” and “Salmon 2020” are projects that aimed/aim to restore Atlantic salmon to the Rhine (Froehlich-Schmitt, 2004). Despite massive river restoration investment along the river and large-scale salmon stocking, there is currently no self-sustaining salmon population and adults can only reach tiny fragments of spawning habitat within their former range (Froehlich-Schmitt, 2004).

The Danube is the largest European river and has the richest fish fauna of all European rivers, providing refuges for retreat during glacial episodes, unlike many of Europe’s rivers. Like the Rhine it has been dramatically modified by channelization, fragmentation, pollution and overfishing (Jungwirth *et al.*, 2003; Schmutz *et al.*, 2014). In Slovakia, 67 fish species were originally found in the Danube and Tisza rivers at the end of the 19th Century (Holčik, 1996). However, due to the industrial development and intensive large-scale agriculture, in Slovakia six species were extirpated by 1994, including beluga sturgeon (*Huso huso*), Russian sturgeon (*Acipenser gueldenstaedtii*), Atlantic salmon and the sea trout ecomorph of *Salmo trutta* (Holčik, 1996). Another 36 fish species were considered under threat and in need of protection (Holčik, 1996). Across 3000 km of river network, more than 650 km had no fish due to barrier construction, river regulation, pollution, arable expansion and water extraction for irrigation (Holčik, 1996). In order to restore aquatic and riparian habitat, large scale habitat and connectivity rehabilitation works were conducted in the Austrian region of the Danube since the 1990s (Schiemer *et al.*, 1999; Schmutz *et al.*, 2014). In addition, ten dams were removed in 2019 in the Danube Delta in Ukraine, to help restore natural hydrological processes.

Fish totally vanished in parts of many rivers within the industrial and heavily populated areas of Great Britain in the second half of the 19th Century and early 20th Century (Maitland and Lyle, 1991). In Scotland, before the 18th Century, Atlantic salmon were present in all accessible reaches of major rivers in the central lowlands (Doughty and Gardiner, 2003). However, in the late 18th Century salmon started to decline in some rivers and the declining trend continued through the 19th Century (Doughty and Gardiner, 2003). By 1900, salmon had become extinct in many rivers where they were formerly abundant (Doughty and Gardiner, 2003). For example, the River Clyde in Scotland and the Thames in England used to have diverse fish communities with about 20-30 species (more in the Thames than Clyde due to its biogeography). Historically, the River Thames held a significant amount of Atlantic salmon (Griffiths *et al.*, 2011). However, the Industrial Revolution and urbanization of London led to increased levels of pollution in the river and salmon was extinct by the 1830s (Griffiths *et al.*, 2011). In the River Clyde, industrial chemical discharges eventually led to an almost complete loss of fishes in the lower reach until the 1980s (Maitland and Lyle, 1991). Although the Thames fish fauna has recovered dramatically since the 1970s (Colclough *et al.*, 2002), due principally to improvements in water quality by pollution reduction (Griffiths *et al.*, 2011), attempts to restore Atlantic salmon have failed due to the large numbers of barriers, diffuse pollution and climate change (Griffiths *et al.*, 2011). Some once common fish species in the Thames such as European river lamprey (*Lampetra fluviatilis*) have failed to recolonise despite water quality now being more than adequate (Lucas *et al.*, 2020).

The River Clyde, used to hold abundant salmon and trout populations before the 18th Century, when salmon were widespread in the system as far as Stonebyres Falls (Doughty and Gardiner, 2003). By the end of the 18th Century, salmon populations were in decline; by the 1840s, there was a strong decrease; by the early 20th Century, salmon had been eliminated from most of the catchment except the River Leven and Loch Lomond system (Doughty and Gardiner, 2003). The decline was due to a combination of factors including dredged estuary habitat, weir obstruction to fish migration and pollution from dye works, chemical works, bleach works, paraffin oil works, distilleries and tanneries (Doughty and Gardiner, 2003). Sewage was also considered to be a major source of pollution in the Clyde and its tributaries. Due to the heavily expanded coal mining industry by the 1920s, effluent from coal washeries also became another serious pollution source (Doughty and Gardiner, 2003). It was not until the 1950s when water quality improved, that fish started to recover in the Clyde (Doughty and Gardiner, 2003).

In England and Wales, most rivers supported a salmon population before the Industrial Revolution, excepting the low-gradient rivers of East Anglia (Mawle and Milner, 2003). Historically, salmon was once so cheap and plentiful that it was a staple food and folklore records that it was not to be served more than three times per week to British servants and apprentices (Hill, 1995). However, during the 19th and 20th centuries, the rapid urban and industrial development caused considerable pollution in rivers and estuaries. The River Thames in southern England was considered to be the first major British river which completely lost its salmon stock, the last record was in 1833 (Mawle and Milner, 2003). Salmon in the River Mersey in Northwest England also totally vanished in the middle of the 19th Century due to pollution (Mawle and Milner, 2003; Ikediashi *et al.*, 2012). In addition, by the 1850s, fish were totally absent from the River Irwell, a major tributary of the River Mersey (Ikediashi *et al.*, 2012). Following growth of industry around the Mersey estuary and expansion of the urban areas of Liverpool and Manchester, it was suggested all fish species were extinct in the main river Mersey by the 1950s (Mawle and Milner, 2003). The River Taff in south Wales suffered pollution from coal mining, metallurgical industries and domestic sewage (Mawle *et al.*, 1985). In 1860, the Commission into Salmon Fisheries mentioned pollution was the major reason for decline of salmon stocks (Mawle and Milner, 2003). Besides pollution, increasing obstruction to migration, water abstraction and overexploitation in fisheries resulted in the loss of fish from many rivers across England and Wales (Mawle and Milner, 2003).

In North East England, both salmon and sea trout have suffered a range of anthropogenic influences and were threatened with extinction in the Tyne, Wear and Tees catchments by the Industrial Revolution. Although emphasis in the literature concerning the impacts of historic river degradation on fishes has often been on anadromous salmonids, a wide range of other species are also impacted, and WFD requires assessment of ecological condition relative to the natural ‘reference’ fish community. To understand the decline and recovery of fish populations in post-industrial rivers, such as those in North East England, where most heavy industry has been lost, it is necessary to know the history of fisheries, mining, industrial development, pollution, water abstraction, and other pressures such as farming, as well as environmental protection actions. An understanding of the impacts of historical human activities on fish communities is valuable. This could shine a light on the factors that have led to biodiversity decline and, loss of ecosystem functioning, and provide an insight for future river management works. Therefore, the major aims of this chapter are to, (1) identify likely contributory factors that led to the functional extinction of the salmon and sea trout in the Tyne, Wear and Tees catchments; (2) summarize the historical timeline of the decline and recovery of fishes and associated environmental quality indicators in the three catchments; (3) evaluate the recovery status of both aquatic habitat and fish biodiversity in the three catchments. This is achieved using evidence from the literature and existing secondary data obtained from governmental agencies, particularly the Environment Agency and its predecessors.

2.2 Materials and methods

2.2.1 Study area

2.2.1.1 River Tyne

The River Tyne is formed by the confluence of two tributaries: the North Tyne and the South Tyne (Figure 2.1), both of which rise in the Pennine Hills. The North Tyne rises in Kielder Forest, it flows south-eastwards for about 63 km then joins the South Tyne at Hexham. In the upper North Tyne valley, the river has been impounded by Kielder reservoir, which is one of the largest artificial lakes in Western Europe. The upper North Tyne and Rede subcatchments are mostly characterised by Kielder Water (reservoir), Catcleugh Reservoir (at the top of the River Rede), commercial forestry and moorland (Environment Agency, 2020a). The South Tyne rises in the Pennine Hills and flows eastwards for about 60 km until joining the North Tyne, then the Tyne flows eastwards for about 58 km until reaching the North Sea at South Shields. The upper catchment of the South Tyne is mostly characterised by moorland. This sub-catchment was seriously affected by metal mining pollution from the start of the 17th Century (Archer *et al.*, 2003). From the confluence of the North and South Tyne to the tidal limit at Wylam, the middle catchment is largely a rural area with productive agricultural land. Several important market towns are located along the river. The middle catchment has a long gravel extraction history since the 19th Century, though gravel extraction from the river corridor itself has been minor in recent decades. The Lower Tyne is dominated by adjacent urban and industrial land. It flows through Newcastle upon Tyne and Gateshead. This area, “Tyneside” was a major shipbuilding and manufacturing centre during the Industrial Revolution. The Tyne Catchment covers an area of 2,943 km^2 and the total river network length is 4399 km. The River Tyne is currently the best salmon river in England (Environment Agency, 2019).

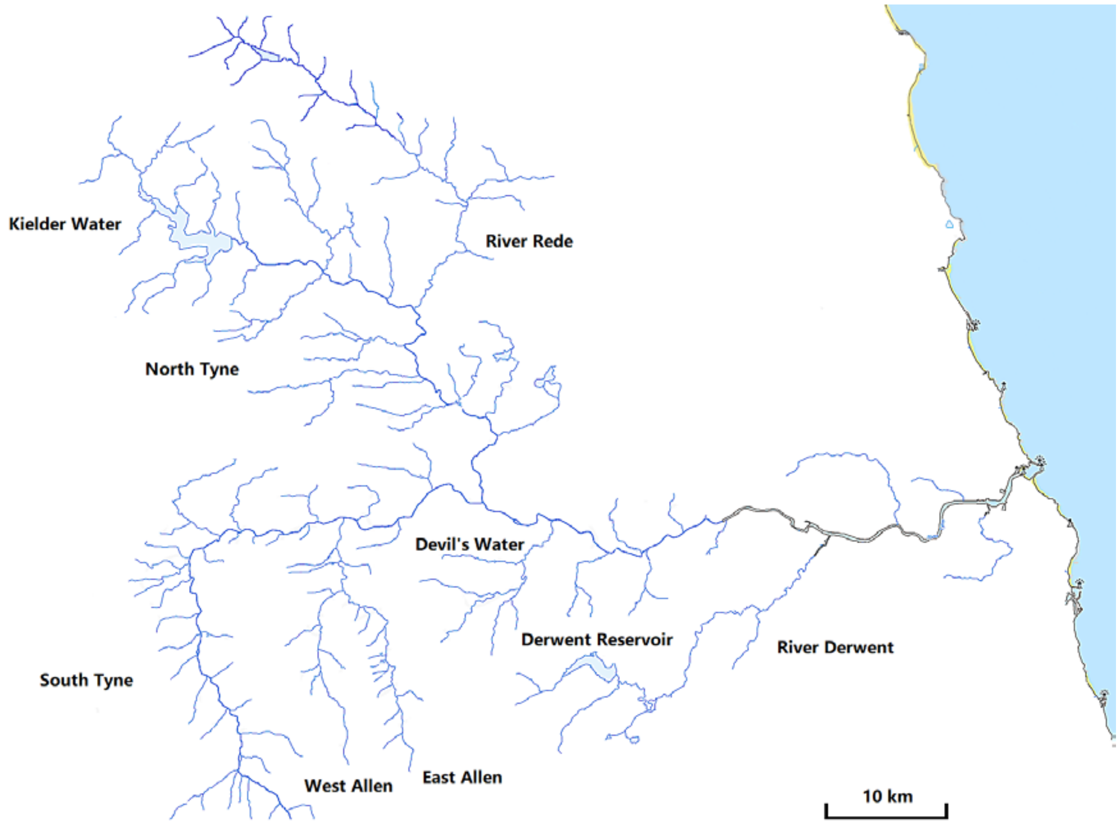


Figure 2.1 The Tyne catchment including major tributaries.

2.2.1.2 River Wear

The River Wear rises in the Pennine Hills and flows eastward for about 110 km until reaching the North Sea at Sunderland (Figure 2.2). The catchment of the upper Wear is mostly characterised by upland moorland (Environment Agency, 2020a). The area is mostly rural and used to be the largest lead-zinc mining region in the world (Kelly, 2002). The landscape of the middle reaches of the Wear is mainly arable farmland, with numerous villages and some larger towns. The middle catchment has a long coal mining, sand / aggregate and shale extraction history close to the river (Neal *et al.*, 2000). Although these mines are now redundant, the catchment is still affected by the legacy of its industrial and mining past in terms of mine water pumping and contaminated water with heavy metals. Pumping of groundwater from mines, particularly coal mines, followed by treatment to reduce pollutants especially metals, is a necessity to protect aquifers and surface water quality (Johnston *et al.*, 2008). In particular, mine water pumping has a major effect on the flows in the Wear's middle reaches (Environment Agency, 2008a). Large amounts of minewater pumping and treatment also occur in the east of Durham to protect Magnesian limestone aquifers for public water supply (Johnston *et al.*, 2008). The lower Wear catchment area is a mix of urban, industrial and arable land. Historically, Sunderland was a major centre for shipbuilding, coal export and glassmaking during the 19th and early 20th centuries.

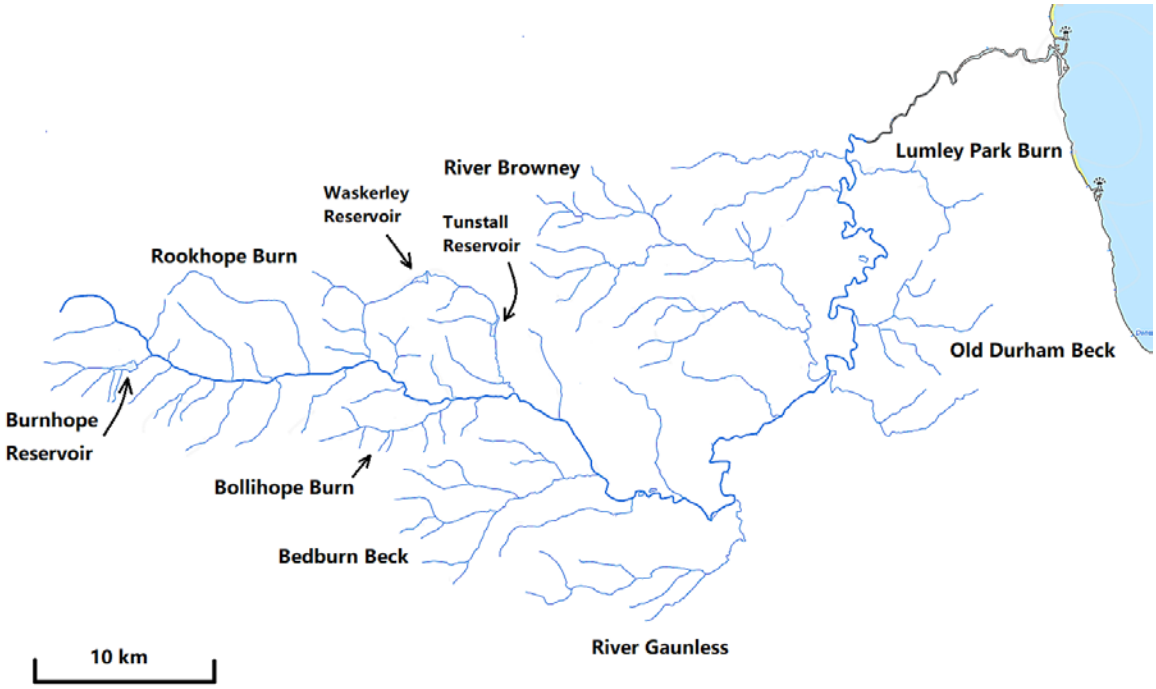


Figure 2.2 The Wear catchment including major tributaries.

The catchment area of the Wear is 1321 km^2 (Environment Agency, 2020a) and the total river network length is 752 km (OS Open Rivers 1: 25,000). Major tributaries of the Wear include the River Gaunless, River Browney, Bedburn Beck, Lumley Park Burn, Rookhope Burn and Bollihope Burn. The tidal limit is at Lamb Bridge between Chester-le-Street and Washington. The Wear is one of the most important Atlantic salmon and sea trout rivers in England (Environment Agency, 2019). The lower Wear suffered severe water pollution from the Industrial Revolution to the 1970s and salmon almost became extinct in the river. From the 1970s onwards pollution sources reduced through the decline of heavy industry and due to better water treatment, the salmon population began to recover, and in recent years the river has had the second highest annual salmon rod catch in England (Environment Agency, 2019).

2.2.1.3 River Tees

The River Tees' source is about 10 km south of the Wear's. The Tees flows eastwards for 160 km and joins the North Sea after passing Middlesbrough (Figure 2.3). The catchment area of the Tees is 1930 km^2 (Environment Agency, 2020a) and the total river network length is 1389 km (OS Open Rivers 1: 25 000). Most of the upper Tees catchment is characterised by upland moorland (Environment Agency, 2020a). Two major waterfalls are located on the main river in the upper reach at High Force and Cauldron Snout. Both waterfalls are complete barriers to fish passage and no migratory salmonids can pass upstream of High Force. Land cover of the middle reaches is mostly categorized as intensive agriculture land. The lower Tees and estuary is largely urbanized as well as having industrialized areas. Industrial activities have been dominated by chemicals and steel making, both of which produce comparatively large quantities of industrial waste (Environment Agency, 2009b). These developed during the 19th Century and at the peak of steel-building there were over 90 blast furnaces within a 10 mile radius of Teesmouth, but the last closed in 2015.

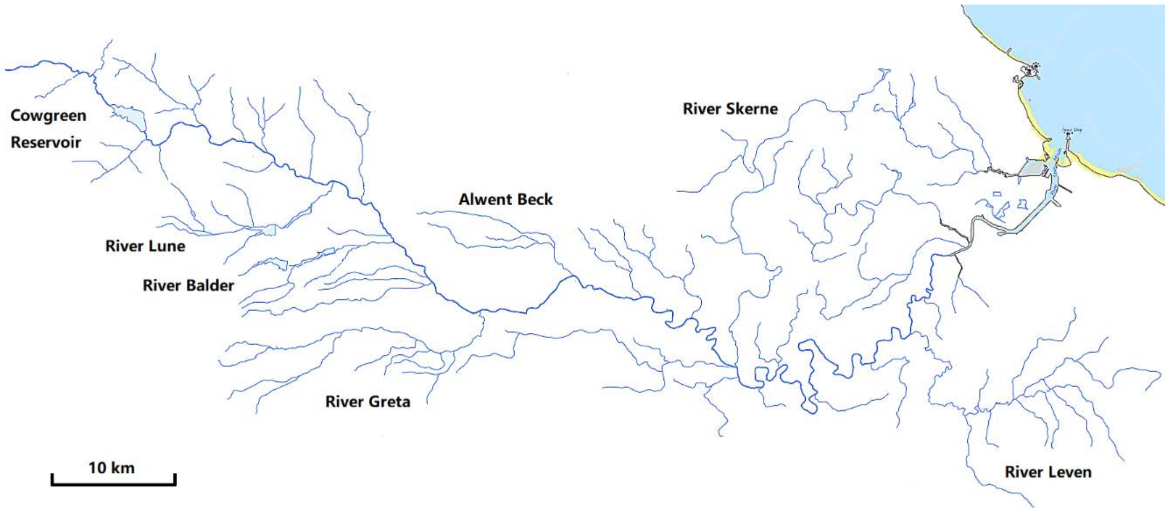


Figure 2.3 The Tees catchment including major tributaries.

The Tees was also a major salmon river until pollution and river barriers caused their decline in the late 19th and early 20th centuries. Major tributaries of the Tees include the Rivers Lune, Balder, Greta, Skerne and Leven. The Rivers Lune and Balder are both isolated from the main river by water storage multiple dams. The Rivers Skerne and Leven were heavily modified for flood defence purposes. A tidal barrage, built 16 km upstream of the river mouth, opened in 1995, in order to limit the tidal movement of polluted water and to facilitate urban redevelopment. Although the Tees Barrage included a salmonid fish ladder in its design, and the water quality of the lower Tees and estuary has improved dramatically in the last 30 years, salmon and sea trout have remained at low abundance by comparison to the Rivers Wear and Tyne to the north (Environment Agency, 2019). The inter-tidal mud-flats of the Tees Estuary are the only substantial areas remaining on the northeast English coast between the Humber 140 km to the south and Fenham Flats 130 km to the north (Smurthwaite, 2006). Despite its industrial development, Teesmouth forms part of the Teesmouth and Cleveland Coast (EU) Special Protection Area and is an international Ramsar site, primarily as it supports 20,000 migrating waterbirds and waders.

2.2.1.4 UK Biodiversity Action Plan species

The UK Biodiversity Action Plan (UK BAP) was published in 1994, and was the UK Government's response to the Convention on Biological Diversity (CBD) (Laycock *et al.*, 2009). It described the biological resources of the UK and provided detailed plans for conservation of these resources. Action plans for the most threatened species and habitats were set out to aid recovery. Fish species including salmon, sea/brown trout and European eel were listed in the UK BAP, and can be found in all three catchments. In addition, the Tyne catchment supports UK BAP species including the white-clawed crayfish (*Austropotamobius pallipes*), pearl mussel (*Margaritifera margaritifera*), otter (*Lutra lutra*) and water vole (*Arvicola amphibius*). The Wear supports UK BAP species including river lamprey, sea lamprey (*Petromyzon marinus*), otter and water vole. The Tees catchment supports UK BAP species including the white-clawed crayfish, otter, water vole and harbour seal (*Phoca vitulina*).

2.2.2 Methods

In this study, historical literature was used to reconstruct, as best as possible, the long-term changes of salmon and sea trout relative abundance in the three catchments and to gather information on the distribution of other fish species as far as is possible. An extensive search of historical literature was carried out for records of salmon and sea trout abundance and captures in North East catchments using a number of different sources including books, newspapers, Environment Agency (and its forerunner organisations) magazines and reports, reports from commissioners and journal articles. Online sources of historical information were mainly gathered from the British History Online digital library, British Library Newspapers Archive, Google Books and UK Government Web Archive. For fish species, search strings including “salmon”, “trout”, “smelt”, “herring”, “shad”, “eel”, “lamprey” etc… were used. For human actions, search strings including “fish”, “fishing”, “net”, “netting”, “catch”, ”dam”, ”weir”, ”lock”, “sell”, ”sold”, ”mine”, ”mining”, ”pollution” etc… were used. For study area, the name of each catchment “Tyne”, “Wear”, “Tees”, and major tributaries “Derwent”, “Rede”, “Kielder”, “Browney”, “Greatham”,”Skerne” etc… were searched. With dates as far back as possible, the information from these sources was cross-validated, to try and ensure accurate information was used.

The long-term annual river rod catches of salmon and sea trout of the Tyne, Wear and Tees catchments, and the long-term trend of Northumbrian coastal net catch data between 1951 and 1990 were gathered from Salmon and migratory trout statistics for England and Wales reports (Russell *et al.*, 1995). Annual rod catch data between 1989 and 1994 were extracted from National Rivers Authority’s Salmonid and freshwater fishery statistics for England and Wales reports (Russell and Buckley, 1991; Russell, 1992; National Rivers Authority, 1993a, 1994a, 1994b, 1995a). Annual rod catch data between 1995 and 2019 were extracted from Environment Agency’s Salmonid and freshwater fishery statistics for England and Wales reports (Environment Agency, 1995, 1996, 1997a, 1997b, 1999, 2000, 2001, 2002, 2003, 2005, 2007, 2008b, 2009b, 2009c, 2013a, 2013b, 2013c, 2014, 2017a, 2017b, 2019, 2020b, 2020c). The catch per rod licence day of salmon and trout data in 1994 was extracted from NRA’s Salmonid and freshwater fishery statistics for England and Wales report. Data between 1995 and 2019 were extracted from EA’s Salmonid and freshwater fishery statistics for England and Wales reports. Besides fish capture data, yearly fish counts for salmon and sea trout data (Tyne: Riding Mill; Wear: Framwellgate Weir; Tees: Tees Barrage) were gathered from the UK Gov website

(Environment Agency, 2020b).

Information concerning the status of salmon and migratory trout populations was contextualised by extracting evidence from books, reports and journal articles concerning the nature of environmental degradation and recovery of the Tyne, Wear and Tees. This was supported by collection of secondary data concerning chemical water quality at tidal and non-tidal locations of each of these rivers and in several tributaries. Archived data (by Environment Agency and predecessors) for these rivers is variable in its timescale and frequency but, in some cases, goes back to the early 1970s for some variables and sites. Therefore, although such secondary data were not available for the main period of decline in water quality of these rivers, some was available for part or most of the period of their recovery. In some aspects, such as diffuse pollution, the catchments may be in a poorer state now than in 1970 (Environment Agency, 2017c). The choice of locations from which to request data, and the range of determinands was therefore influenced by the likelihood of these being the most useful indicators of pollution impact on fishes and of being most likely to be recorded for the highest proportion of samples. From these, a more limited range of sites were chosen for presentation in this thesis, including where possible, tributaries examined in Chapters 3-5 of this thesis. The primary aim of this chapter was to provide a descriptive (often using textual evidence) rather than statistical investigation of historical patterns. Such approaches are common within the social sciences, less so in the natural sciences, and reflect the limited quality assurance of some information sources, particularly from pre-1960 sources. The most reliable data sources, on which some statistical analysis could be applied, were considered to be water quality data archived by the Environment Agency, in some cases dating back to 1973. Although it is known (based on reports from historic documents) that basic key water quality data (e.g. dissolved oxygen) for various key sites (e.g. Tyne estuary) were recorded weekly or monthly in the 1970s and 1980s, some of these raw data could not be obtained from the Environment Agency's digital database, despite repeated attempts and requests. Some of these data might be archived paper records, but repeated attempts to access such data were unsuccessful. Linear regression was used to analyse the long-term key water quality parameters (DO, BOD, ammonia, nitrate, orthophosphate/ phosphorus, zinc and lead) at key sites. Before analysis, data were checked for normality test, and log (x+1) transformations were applied when needed.

In addition, the EA Freshwater fish surveys (National Fish Populations Dataset) dataset was used to assess the long-term fish community changes at major tributaries in the Wear and Tees catchments from the 1990s onwards (Environment Agency, 2020c), these being the catchments in which empirical work was undertaken in Chapters 3-5. These data are presented for descriptive interpretation and not subjected to statistical interpretation, due to inconsistent samping periodicities and changes in recording methodologies over the timescale of interest. For the Wear catchment, the Cong Burn, River Browney (including River Deerness) and Bedburn Beck sub-catchments were chosen. The first two of these are degraded streams in which multiple connectivity restoration works have been carried out in recent years, while Bedburn Beck is a relatively natural stream with fewer artificial impacts (see Chapter 5 for details). For the Tees catchment, the Low Moor site in the main river, Claxton Beck (upper part of Greatham Creek), River Skerne, Clow Beck and River Lune were chosen. All sites have the longest fish survey histories in the Tees catchment. The River Skerne represents a degraded stream joining the river’s middle reach, which was historically heavily modified and polluted. Clow Beck represents a cleanish stream joining the river’s middle reaches, but for which in the last 50 years neighbouring land has become increasingly intensively farmed. The lower reach of the Lune represents a high water quality upper reach stream (albeit impounded upstream), two sites located in the lower River Lune were selected for data analysis. Fish data for these sites in the Wear and Tees were available only as far back as the early to mid-1990s – although regular electric fishing surveying in these rivers occurred at least a decade earlier (M. Lucas, pers. comm) it appears that these data have not been archived effectively and may no longer exist, despite unsuccessful efforts to locate any remaining.

2.3 Results

2.3.1 River Tyne

2.3.1.1 History of Tyne salmon

Atlantic salmon used to be very abundant in the Tyne. During the Middle Ages, fishermen used a succession of weirs for capturing salmon and other fishes, including in North East English rivers such as the Tyne; seine nets were used to catching fish below the weirs (Champion, 2003). In the tidal area, groynes were installed in such a way as to deflect migrating salmon into fixed nets or traps or into shallow water where fish could be netted or gaffed (Champion, 2003). Around the 1760s, a condition was inserted in all indentures in Newcastle that the apprentice was not obliged to eat salmon above twice per week (Bewick, 1862). Records have shown that salmon netting and trapping in the Tyne was the major local fishing industry until the end of the 19th Century when marine fisheries (e.g. herring *Clupea harengus*, cod *Gadus morhua*) became increasingly important. It was reported on 12th June, 1755, that over 2400 salmon were taken in the Tyne; on 20th June 1758 more than 2000 salmon were taken in the river and on 6th August 1761, 260 salmon were caught in one draught (one net haul) at Newburn (Mackenzie, 1827). Also a few records gave evidence that people fished for salmon as a sport. Thomas Bewick mentioned his grandfather (around 1700) was an expert salmon angler on the Tyne and other local rivers, in his memoir (Bewick, 1862). However, there are no records for sea trout fishery captures until the 19th Century (Champion, 2003). This is partly because sea trout were not always distinguished from salmon in early records, or were not deemed worthy of records. From the 19th Century, the coastal net fishery for salmon developed greatly, and large numbers of fish (salmon and sea trout) were taken each year (Champion, 2003). However, these coastal fisheries intercept salmon and sea trout moving along the coast to a range of rivers and so do not necessarily reflect catches allied to a particular nearby river or even a group of nearby rivers.

2.3.1.2 Pollution of the Tyne

The first environmental challenge for the Tyne fish populations was probably the impacts of water pollution. The lead mining history in the North Pennines can be traced back to the Middle Ages (McParlin, 2011). Large scale mining started in the early 17th Century in the Tyne catchment (Archer *et al.*, 2003). Following the development of mining, ore separation and smelting technologies, the northern Pennines region (including south Tyne, upper Wear and upper Tees catchment) became Britain's most important lead and zinc mining areas by the 18th Century. All three rivers were seriously polluted due to the mining activity. The production of lead from this area peaked around the 1850s and zinc production peaked around 1900 (Archer *et al.*, 2003). In the Tyne catchment, the majority of lead and zinc mines were located in the South Tyne and River Derwent area. In the River Derwent, the headwaters suffered serious pollution from the lead mines. The middle reaches of the Derwent were also affected by coal mine drainage at Blackhall Mill and the Derwent-Tyne confluence was polluted by the discharge from Consett steel works (Archer *et al.*, 2003).

During the mineral processing activities, large amounts of lead, zinc, and related elements such as cadmium and copper were released into the river system. A mining method called “hushing” was developed in the Northern Pennines and Yorkshire Dales region, it was carried out by damming a stream near suspected metal ore veins, then releasing the water suddenly to wash the surface soils away to reveal the bedrock and expose the underlying metal ores (Hudson-Edwards *et al.*, 2008). During the “hushing” exploration, large amounts of soil would be washed into the river. The discharges from the developed mine could carry metal-rich sediments, and these sediments would be transported downstream and deposited on banks and floodplains and reworked during successive flood episodes (Archer *et al.*, 2003; Hudson-Edwards *et al.*, 2008). The degree to which heavy metals are dissolved in water and therefore bioavailable and toxic depends on water pH, and so on the natural buffering capacity of the surrounding area (Kelly, 1988). In particular water arising from mine adits is often rich in iron, zinc or copper sulphide which is oxidized, due to air contact, to sulphates of these metals, with associated H^+ ion production. Although large parts of the North Pennines have limestone outcrops, limiting the tendency for acidification and dissolving of heavy metals, some areas such as the South Tyne have lower buffering capacity and were more susceptible. In 1870, salmon in the South Tyne nearly vanished due to pollution from lead mines around Alston (Marshall, 1992). In 1891, the Clerk of the Fishery Board gave a supposition that gravel abstraction from the river would expose and spread the toxic sediment out, and result in the death of breeding salmon (Champion, 1991). Dissolved lead is highly toxic to eggs and larvae of salmonids and other fish (Jezierska *et al.*, 2009; Lee *et al.*, 2019); it has no natural biological activity and is bioaccumulated, causing neurotoxic effects in vertebrates (Lee *et* *al.*, 2019).

Apart from metal mining, the coal mining and trading also caused pollution issues to the Tyne catchment. The major problem of coal mining was the waste water from coal washeries and mine dewatering, as well as the presence near many coal mines of coking plants. Coal washing caused covering of the bed with large amounts of inorganic particulates, sufficient to fill gravel interstices and smother the benthic ecosystem, reducing hyporheic flow and causing substrate deoxygenation. Suspended solids (particularly flocculated ferric oxide material, if water pH is near neutral or weakly acidic) from coal mine drainage were deposited in the river channel, covering the riverbed and destroying the natural benthic community. Such pollution is usually referred to as “ochre” and is a common outcome of heavy metal and iron mining. Sub-surface mining often progresses below the water table, so during normal mine activity, the water needs to be pumped out to keep the water level below the working area. After the mine is closed, if the pumping ceases, then water fills the shaft tunnel and is then discharged to the river again. As stated above, the resulting ‘acid mine drainage’ can have major effects, particularly where there is little buffering capacity in the receiving water, but even where buffering is sufficient, ochre pollution causes immense damage until the release of oxidized metals is exhausted or treated. During floods, significant amounts of sediment-borne heavy metals are washed along rivers and can be temporarily stored in the gaps between gravels (Macklin and Lewin, 1989). The toxic sediment deposition can pose a potential threat to both fish and water quality. The phytotoxic effect of contaminant metals on riparian vegetation is also considered to be a major factor of causing bank instability and the slow recovery of many mining-affected river systems (Hudson-Edwards *et al.*, 2008).

Because coal (and metals) was available in localized seams, ‘pit villages’ sprang up in the region of these to exploit the production of coal, resulting in numerous sources of mine pollution. During the 19th Century many pit villages also developed coke works at which coal was processed to coke, used in the steel making process, such as at Consett. Coke production generates large amounts of polluting material including polycyclic aromatic hydrocarbons (PAHs), benzenes, phenols, quinolines and ammonia (Liu *et al.*, 2017), significant amounts of which were leached or washed into waterways and were toxic to fishes and other biota (Baumann and Harshbarger, 1995; Liu *et al.*, 2017). Waste 'slag' heaps from this process continued to act as a pollution source for leachate into nearby freshwaters. The pit villages also produced their own sewage, released untreated into streams, or leaching from middens of 'night soil'.

From the beginning of the 19th Century, followed by the rapidly increased industrialisation, the salmonid fisheries in the Tyne faced another challenge. The Tyne estuary was developed with coke, iron, chemical works, tanneries and a wide range of other heavy industry and transportation (Linsley, 2003), by the mid-19th Century creating a large, concentrated area of intense pollution. These industrial works caused significant pollution in the river. In addition, metal mining activity also seriously degraded the water quality and increased the quantity of fine sediments getting into the river channels. The population of Newcastle increased from 27,500 to 128,500 from 1811 to 1871 (Archer *et al*., 2003). Along with the population increase, the disposal of domestic waste water became another issue for the Tyne estuary in the 19th Century. Slowly, more homes were connected to the sewerage system and, because no treatment of this sewage occurred, the level of oxygen-demanding input to the estuary increased progressively (Archer *et al*., 2003). Crucially also, rather than the waste being diluted and dispersed out to sea as presumed by the engineers, oxygen demanding waste was retained for long periods in the estuary due to the tidal nature of the river (combined with salinity stratification), with a net downstream movement of just 400 m per tidal cycle (Archer *et al*., 2003). This resulted in severe oxygen depletion and ammonia elevation and coating of the estuary bed with toxic sediments. In 1891, the Clerk of the Fishery Board mentioned salmon smolts getting killed by pollution in the estuary in evidence given to a Royal Commission (Champion, 1991). In 1895, the dry spring reduced the flow in the estuary, and pollution resulted in thousands of salmon smolts dying when negotiating the brackish waters (Marshall, 1992). Furthermore, the salmon population was functionally extirpated in the South Tyne due to the pollution from lead mining (Champion, 1991). From the late 1800s the rod fisheries in the upstream reaches of the Tyne started to decline, although it seems likely that some of the Tyne weirs had impacted salmon populations to some degree before this (see section 2.3.1.3

below). The total rod catch of salmon in the Tyne reduced from 3201 in 1885 to 125 in 1898 (Champion, 1991). In the 1930s, salmon sold in the Newcastle market were found to have a tainted tarry flavour, and complaints were received from London fish merchants due to the engine oil taste from Tyne salmon (Champion, 2003; McParlin, 2011). This was probably caused by the phenols discharges from Derwenthaugh and Norwood coke works (Champion, 2003).

In 1959, the Board of Conservators for the Fishery District of the River Tyne reported that large numbers of smolts were killed when entering the tidal reach during the downstream migration (Champion, 1991; Marshall, 1992). And 1959 was the first year that there was no reported rod catch of salmon (Champion, 1991). Although this could imply a lack of recording, more likely it reflects minimal fishing due to negligible numbers of salmon making rod fishing not worthwhile and with none caught. In 1960, there were nearly 275 major discharges of untreated waste water drains to the estuary (Archer *et al.*, 2003). Although The Rivers Pollution Act of 1976 made it an offence to discharge untreated pollution into an inland watercourse this was neither well applied, nor applicable to estuaries. There was no effective legislation on control of the discharge to the estuary until the Rivers (Prevention of Pollution) Act of 1961 and Control of Pollution Act in 1974. However, by that time, the Tyne salmon population was already functionally eliminated by the pollution (Champion, 2003). The degree to which salmon recolonized from residual Tyne genetic stock, by strays from adjacent rivers or by Kielder stockings is considered briefly later (section 2.3.1.5). In the early assessment, it was estimated that an overall hatchery (1980 – 1986) contribution of up to 20% to the cumulative adult run, would have played an important role in accelerating and stabilising stock recovery in the Tyne in the early recovery stage (Milner *et al.*, 2004).

2.3.1.3 Connectivity and habitat alteration in the Tyne

Following the increased demand for concrete in construction in the 1930s, gravel and sand extraction across the UK rapidly increased. The commercial production of gravel increased from 2.2 to 106 million tonnes between 1922 and 1968 in the UK (Archer, 2003a). In the Tyne catchment, at least 15 historical gravel extraction sites directly linked to the river channel (rather than on the alluvial plain outside normal flood reach of the river) have been identified. Ten gravel extraction sites were located on the main river and the remaining five works were located on the south Tyne (Archer, 2003a). These extraction works covered nearly 20 km total river distance, and it is estimated that 4.6 million tonnes gravel were removed from the river channel between 1891 and 1972. Gravel extraction has had several negative impacts on both in-stream habitat and aquatic species. It can deepen the river depth at the working site, increase upstream erosion and channel widening and lead to the reduction in spawning grounds for lithophilous (gravel-cobble preferring) fishes such as salmon, trout, lampreys and minnows (*Phoxinus phoxinus*).

Besides gravel extraction, in-stream barriers such as dams and weirs also posed significant impacts on the river. Historically, numerous weirs have been constructed in the Tyne catchment. These weirs served different purposes from providing water for mill operation to fish trapping. There are two weirs considered to be a main reason for the decline in Tyne salmon catches before 1800. One weir located at Bywell on the main Tyne, and a second weir (Derwenthaugh Dam) located at Winlaton Mill on the River Derwent, prevented fish from passing up the river to breed, particularly during low flows (Mackenzie 1827; Champion, 2003). The construction of the weirs also benefitted poachers; it was recorded that more than 200 salmon were taken on one occasion in the pool below Bywell weir during the closed season (Marshall, 1992). Apart from these two weirs mentioned above, the Warden weir on the South Tyne, and Woodburn weir on the River Rede also posed significant impacts on fish migration (Marshall, 1992). In 1862, Bywell weir was removed due to the complexity of building a fish pass on it. In 1870, Woodburn weir was removed, and sea trout were observed upstream in the River Rede for the first time since its construction (Marshall, 1992). In 2012, a Larinier super active baffle fish pass and an eel pass were installed on the Derwenthaugh Dam in the River Derwent. In the main River Tyne, below Hexham Bridge, gravel extraction seriously impacted the river bed. To protect the bridge foundations from riverbed erosion, a series of weirs were constructed underneath the bridge in the 1950s (Champion, 1991). Since then, the weirs at Hexham have become a major obstruction for the migratory salmon and sea trout heading upstream to spawn. Only recently (2015) has a large upstream fish pass, suitable for the size of the structure been added at Hexham Bridge for migratory salmonids, although this is unsuitable for other migratory species such as lampreys and eels.

Multiple impoundments in the Tyne catchment have altered hydrology and prevented access to spawning and nursery streams in the upper reaches. Whittle Dene, Hallington, Colt Crag and Catcleugh Reservoirs were all built in the 1800s for water storage and treatment for the Newcastle-Gateshead conurbation (Figure 2.4). Further demand for water supply by the mid-20th Century led to a new phase of impoundment in the Tyne (Archer, 2003b). After 1950, most pollution sources in the upper River Derwent had disappeared. Then, in 1957 the construction of Derwent Reservoir was proposed in order to provide sufficient water supply for Gateshead. Work started on building the reservoir in 1960 and it opened in July 1967. After the completion of the Derwent Reservoir dam (Figure 2.4), without any fishway, the upstream river reach became totally inaccessible to all migratory fish species. Since the large numbers of weirs downstream precluded access near the bottom of the river anyway, access past Derwent dam was not considered an issue. After the construction of Derwent Reservoir, in the late 1960s plans were developed for another reservoir to meet the predicted further demand of both industry and domestic water usage (Archer, 2003b). The Kielder scheme began in 1975 and was completed in 1981 (Figure 2.4). Kielder reservoir impounded 11 km of salmon and trout spawning grounds and prohibits upstream access to these. Gravel in these tributaries had been extracted already, so they might not be considered as high-quality spawning habitat, but the construction of Kielder dam still had a significant impact on the potential for salmon recovery. As part of the legislation, since Kielder was built with no fishway, annual stocking of juvenile salmon/eggs was required as a mitigation (see section 2.3.1.5 below).

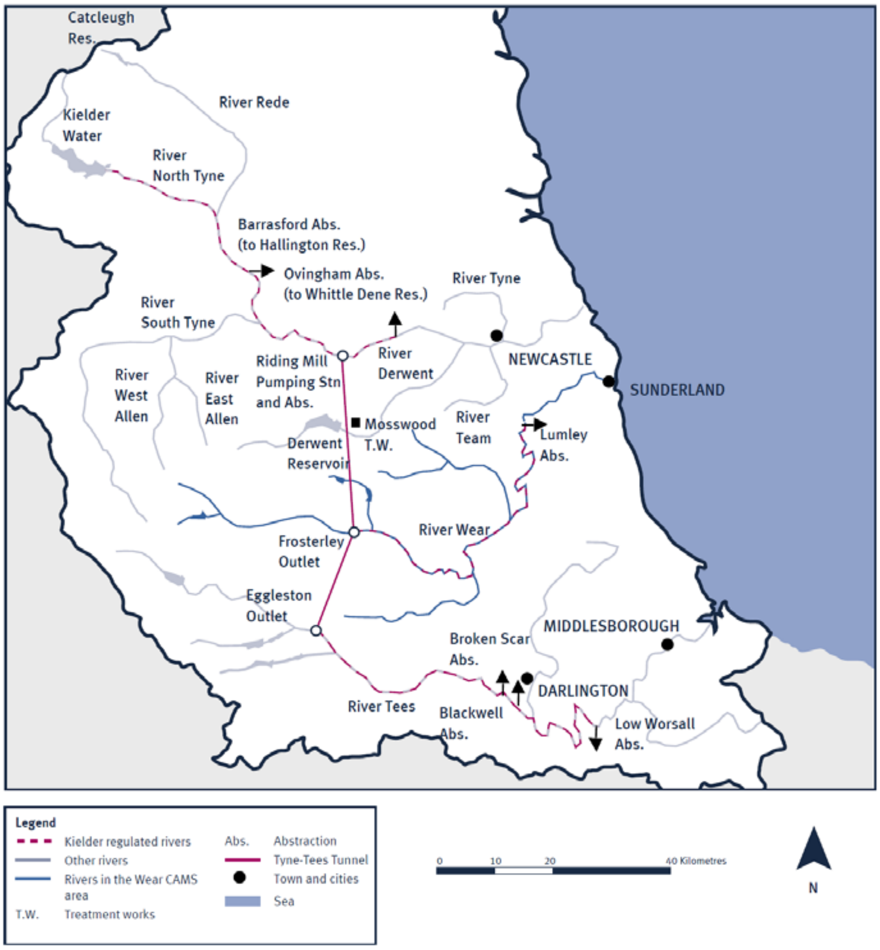


Figure 2.4 The Kielder Water Scheme and regulated river reaches. Also shown are Derwent Reservoir and several Wear/Tees reservoirs, as well as the Tyne-Wear-Tees water transfer pipeline. Source: Environment Agency (2006).

Kielder reservoir was constructed in order to regulate the rivers Tyne, Wear and Tees (Figure 2.4; Environment Agency, 2006), in order to facilitate abstraction for water supply on these rivers, and to augment river flow during dry periods due to abstractions. The heavy industry at Teesmouth, in particular, was expected to require large amounts of water, potentially beyond the levels of supply that the Tees could provide. At Riding Mill, 56 km downstream of Kielder dam, a pumping station and associated weir was built at the mill in 1982 (Cave, 1985). A 38 km long pipeline (Figure 2.4) was built between Riding Mill, Rogerley (River Wear) and Eggleston (River Tees) to facilitate the inter-river transfers (Cave, 1985). The scheme has maintained water supplies to the region, during drought years it delivers water to downstream reaches for abstraction, domestic and industrial use. In addition, rivers such as the Tyne, Wear and Tees can be kept at minimum levels to meet ecological needs even when rainfall has been low. This has enabled Northeast England to avoid water use restrictions that were experienced in many other parts of the country (Environment Agency, 2006). In addition, regulated water can be used to mitigate serious pollution events in the Rivers Tyne, Wear and Tees (Environment Agency, 2006). However, none of the major impoundments in the Tyne catchment have fishways and the impoundments also alter the chemical conditions and hydrology of the constituent rivers as well as the receiving rivers.

An upstream fish pass was constructed on the left side of the weir at Riding Mill in 1982. It has three chambers which are connected by submerged orifices. A temporary timber baulk pass was built on the middle of the weir in November 1982, in an attempt to pass fish upstream over the structure during high flows. However, the timber fish pass was unsuccessful and was removed in May 1983. An adult salmonid fish counter was installed at Riding Mill in 1996. In the late 20th Century, a salmonid fish pass was installed on the weir at Chollerford on the North Tyne. Chollerford is also the second of two adult salmonid fish counters on the Tyne, 16 km upstream of Riding Mill. It does not provide a whole river estimate of run size. In the early 21th Century, a salmonid fish pass was built on Haltwhistle Alston Arches on the South Tyne and another salmonid fish pass was installed on Dilston Weir on the Devil's Water (Environment Agency, 1999b). Fishways were built on several weirs in the lower Derwent in the 2010s, these having more nature-like characteristics and suited to a wider range of fish species (Tyne Rivers Trust, 2020).

2.3.1.4 Recovery of the Tyne water quality

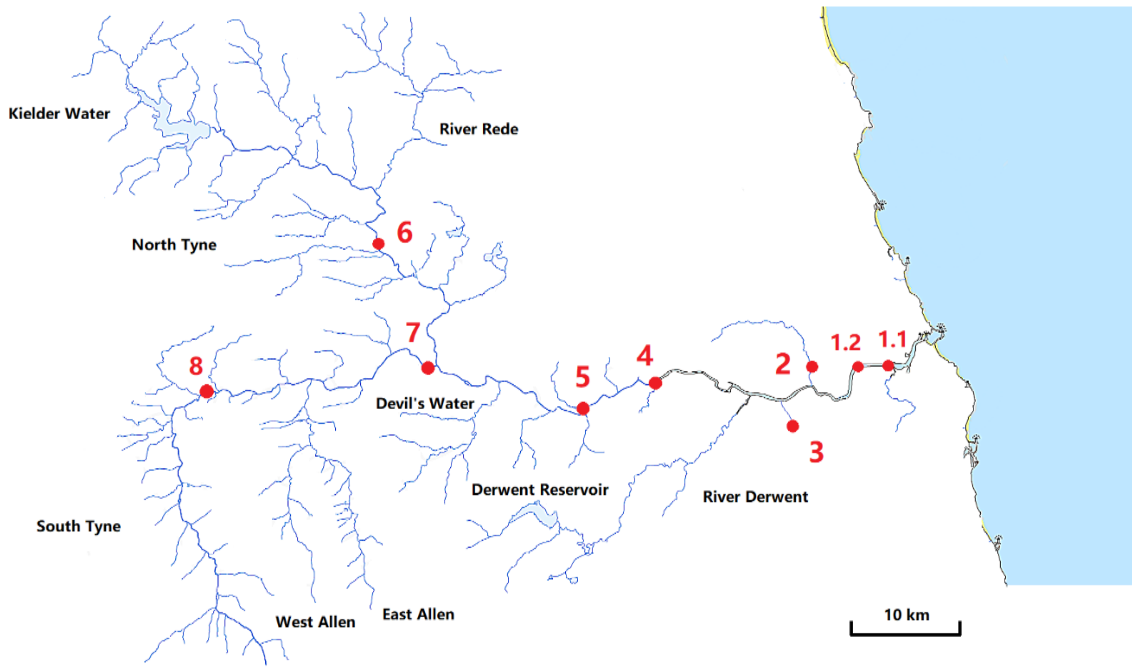
Lead mines in the North Pennines region began to close in the early 20th Century. Followed by the reduction of mining, the water quality in the upper reaches of the rivers had a notable improvement (McParlin, 2011). Metal mines were finally closed in the early 1930s, and the coke works (and a large amount of related heavy industry, particularly on Tyneside) in the Tyne area declined and finally closed in the early 1980s. There have been two main periods of improvement of water quality improvement in England, including for rivers such as the Tyne. The first main period was during the 1960s and 1970s (Mawle and Milner, 2003). In 1958, 20 local authorities formed a working party to investigate measures to reduce the pollution of the Tyne estuary and adjoining beaches (Ord, 1988). The Tyneside Joint Sewerage Board was constituted in 1966 to promote the Tyne Sewage Treatment scheme which was inherited by Northumbrian Water Authority in 1974 who further developed the scheme (Ord, 1988). The Howdon treatment work scheme was first recommended in 1964, comprising a sewer network to be constructed along both bank of the Tyne to catch more than 200 outfalls, with waste water transported to a new sewage treatment plant at Howdon (Foster, 2003). Construction of the Howdon Treatment Works started in 1973. The sewers were constructed along both banks of the Tyne for more than 35 km, between Ovingham and Tynemouth. The overall interceptor sewer was commissioned in late 1983. Visual pollution, oxygen demanding waste and smell in the Tyne estuary significantly reduced after the work was completed, but waste water, following secondary (activated sludge) treatment, is still returned to the estuary, rather than to the open sea (Foster, 2003). Historically this has still meant a tendency for oxygen-demanding waste (albeit at a much lower level) to be retained within the estuary, due to tidal action, especially during periods of low river flow, particularly during summer.

Followed by the reduction of pollution in the estuary, the numbers of returning anadromous salmonids showed an increasing trend, reflected in increased rod catches in the 1980s (see section 2.3.1.5 below). Surveys of demersal fish in the Tyne, Tees and Wear estuaries began in 1981, in order to assess the effects of pollution and its reduction by various control measures (Pomfret *et al.*, 1988). From 1982 to 1988, increased numbers of fish species were found during beam trawling in the Tyne estuary (Pomfret *et al.*, 1988). A significant increase in flounder (*Platichthys flesus*) catches was recorded at three stations in the upper Tyne estuary (Pomfret *et al.*, 1988). Between April 1981 and March 1982, the annual average DO was 7.1 mg/L and the lower 95%ile was 0.6 mg/L from the tidal Tyne upstream of Derwent confluence (Blaydon area) (Hugman *et al.*, 1984). A study of catch data from the Tyne estuary (provided by the Northumbrian Water Authority) showed that salmon passed upstream when the DO was between 4.5 and 6.8 mg/L (Hugman *et al.*, 1984), although salmon also migrate at higher DO's when these are available during their migration periods. However, during the summer periods, the low dissolved oxygen levels in the estuary reach still caused fish kills and the problem was persistent through the 1990s. In 1990, water quality in 30% of the Tyne estuary was classified as ‘poor’ (Mawle and Milner, 2003). Salmon that remained in the cool estuary, particularly during lower summer flows with warm river water, often became stressed and susceptible to pathogenic infections (National Rivers Authority, 1994c, 1995b). Apart from salmon, flounder was found to be less abundant or even absent during beam trawling at upper Tyne estuary stations during the summer season (Pomfret *et al.*, 1988).

The second period of water quality improvement followed the privatization of the water industry in 1989, since when there has been stronger regulation of water quality, by the National Rivers Authority and subsequently the Environment Agency (Mawle and Milner, 2003). However, in the lower Tyne estuary, water quality was still affected by sewage discharges. In the Howdon and Hebburn area, decline of fish abundance and the number of species was observed during beam trawling survey between 1982 and 1996 (Gill *et al.*, 2001). This suggested a general decline in water quality in the lower estuary, due to increased discharges from the Howdon STW (Gill *et al.*, 2001). In the summer of 1994, it was estimated 500 fish died in the Tyne estuary mostly larger salmon and some smaller sea trout (National Rivers Authority, 1994c). In the summer 1995, it was estimated 2000 fish including more than 900 salmon and sea trout died in the estuary due to low dissolved oxygen concentrations and related stressors (National Rivers Authority, 1995b, 1995c; Environment Agency, 1996b).

To monitor water quality under the Water Resources Act 1991, the National Rivers Authority (NRA), introduced a method to examine separate stretches of estuary and freshwater in terms of their chemical, biological, nutrient and aesthetic qualities (Environment Agency, 1996c). This method was called the General Quality Assessment (GQA) scheme. Rivers were first assessed using this scheme in 1995 and it ran until 2009. The scheme consisted of four main parameters of measurement which were: chemical quality, biological quality, nutrient status and aesthetic quality. After 2009, the GQA was replaced by metrics underpinning the Water Framework Directive (see Chapter 1). However, these metrics are not cross-convertible and so of limited use for indicating long-term changes.

In the Tyne catchment, nine sites were chosen to examine the long-term chemical trends (Figure 2.5). Two sites were located in lower catchment tributaries and remaining sites were located in the main river including North Tyne and South Tyne. One site located in the Ouse Burn, which is a small tributary running from the north through Newcastle and this river was historically polluted (Archer *et al*., 2003). The River Team is a small tributary that meets the Tyne estuary on the right bank and passes through an urban area which was subject to heavy industry in the past and historically this river was badly polluted. Both tributaries reflect the trend of water quality in tributaries running through built up or industrial areas in the Tyne catchment.



1.1 Tyne at Jarrow
1.2 Tyne at Hebburn
2. Ouse Burn at Jesmond Dene
3. Team at Third Avenue Bridge
4. Tyne at Wylam Bridge
5. Tyne at Bywell
6. North Tyne at Wark
7. South Tyne at Warden Bridge
8. South Tyne at Haltwhistle

Figure 2.5 Tyne catchment and sites with long-term chemical trends for which data are presented in this thesis.

The Tyne estuary water quality data series is crucial in understanding water quality improvement, the earliest recoverable data (at Hebburn and Jarrow sites) were since

1969 (Ratasuk, 1972), and the earliest recoverable data at the tidal limit (Wylam Bridge) were since 1974. In the Tyne at Hebburn, low DO values of ~3.7 mg/L were recorded in 1970 (Ratasuk, 1972), and low DO values of ~4 mg/L were recorded between 1994 and 1997. These are at levels which are very stressful to juvenile and adult salmonids and can cause mortality if exposed for to these levels for several days, particularly in warm weather (Alabaster & Lloyd, 1982). Oxygen levels in the tidal Tyne increased and became relatively stable after 1997 with summer minima not reducing below ~6 mg/L, although there was a significant increase between 1991 and 2020 (Figure 2.6; Table 2.1). Peak values of ammoniacal nitrogen, orthophosphate, zinc and cadmium appeared in the early 1990s, then reduced and became more stable after that, with ammonia and orthophosphate both exhibiting significant declines for available periods of data (Table 2.1). Other chemical components did not show any clear trends.

In the Ouse Burn, data were available from 1973, but halted in 2006. The pH varied markedly during the late 1980s, then became stable around 8 after that (Figure 2.7). Dissolved oxygen, BOD, ammonia, zinc and lead all decreased significantly, and orthophosphate increased significantly, over the periods for which data were available (Table 2.1). Biochemical oxygen demand (BOD) showed large variations prior to 1982 but has become more stable and generally lower since then (Figure 2.7). Ammoniacal nitrogen concentration dramatically reduced after 1992 (Figure 2.7). Both lead and zinc concentration were at high levels in the 1980s, then significantly reduced after that and became relatively stable (Figure 2.7).

In the River Team, with data spanning 1973-2019 (but differing across determinands) clear decreasing trends of BOD, ammonia, lead, zinc, mercury and cadmium levels were observed (Figure 2.8). For the subset of determinands statistically analysed, BOD, ammonia, lead, zinc and phosphorus decreased, while oxygen increased significantly (Table 2.1). The BOD showed a decreasing trend since the early 1990s then became relatively stable after that (Figure 2.8). Nitrate concentration peaked several times in the late 1980s, no extreme value was detected after that. Ammoniacal nitrogen dramatically reduced after 1993, from when the value has been maintained below 10 mg/L (Figure

2.8). Both mercury and cadmium concentration dramatically declined in the late 1980s and early 1990s and have remained at low levels since (Figure 2.8). Other chemical components did not show any clear trends during the sampling periods.

In the Tyne at Wylam Bridge (data, 1974-2020 but dates varying across determinands) BOD, ammonia, nitrate, phosphate, zinc and lead all decreased significantly and oxygen increased significantly (Table 2.1). Ammoniacal nitrogen concentrations varied greatly between 1974 and 1992, but became relatively stable and low after that (Figure 2.9). The BOD concentrations varied widely between 1974 and 1995, then declined and became more stable after that (Figure 2.9). Phosphate also showed a gradual decline over the timescale, but was never particularly high. Peak values of lead, zinc, iron, mercury and cadmium appeared in the late 1970s and early 1980s, then reduced and became more stable after that (Figure. 2.9). Other chemical components did not show any clear trends during the sampling periods.

In the Tyne at the Bywell (data, 1973-2019 but dates varying across determinands), BOD, ammoniacal nitrogen, orthophosphate, lead and zinc all showed a decreasing trend (Table 2.2). For BOD, ammonia and orthophosphate this was especially apparent after 1990; BOD and ammoniacal nitrogen concentration were maintained below 10 mg/L after that (Figure 2.10). The decrease in lead at about the same time appears to reflect a sudden transition in detectability in analyses, although this is unconfirmed. Other chemical concentrations did not show any clear trends during the sampling periods.

In the North Tyne at Wark (1974-2006, date range varying across determinands), zinc and lead levels decreased significantly (Table 2.2). The zinc concentration decreased since the early 1990s, although this looks suspiciously like being due to an increase in detectability at low concentrations, as the baseline suddenly decreases (Figure 2.11). For pH, BOD, ammoniacal nitrogen and nitrate concentrations, a few peak values were observed in the late 1980s, but the overall trends were relatively stable. Other chemical concentrations did not show any clear trends during the sampling periods.

In the South Tyne at Warden Bridge (1973-2020, date range varying across determinands), ammonia, total phosphorus and lead levels decreased significantly (Table 2.2). For pH, BOD, ammoniacal nitrogen and nitrate concentrations, a few high values were observed in the late 1990s, but the overall trends were relatively stable (Figure 2.12). Total phosphorus concentration varied between 1974 and 2007, then declined and became relatively stable after that (Figure 2.12). Zinc, though variable, tended to be at quite high concentrations of about 100 ug L^{-1} over the entire period of records. For lead concentration, a few high peaks were observed between 1977 and 2007, then it became relatively stable (Figure 2.12). Mercury and cadmium concentrations were varied greatly between 1975 and 1991, then largely declined and became relatively stable after that.

In the South Tyne at Haltwhistle (1979-2020, date range varying across determinands), ammonia, nitrate, orthophosphate, zinc and lead levels all decreased significantly (Table 2.2). A few high values were observed in BOD, ammoniacal nitrogen and nitrate concentrations in the late 1990s, similar to the Warden Bridge site (Figure 2.13). Lead, zinc and iron concentrations were variable between 1979 and 1994, then became relatively stable after that, although two high vales were observed in both lead and iron concentrations in the early 2000s (Figure 2.13). Mercury concentration varied between 1980 and 1993, then declined and became relatively stable (Figure 2.13). Orthophosphate concentration varied between 1979 and 1982, then declined and became relatively stable after that (Figure 2.13).

Table 2.1 Linear model summaries of changes in key water quality parameters in the Tyne catchment (S1-S4). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea.

Site	Parameter	Periods	*df*	*t*	*P*
1	DO	1991-2020	1,122	2.49	**0.014**
Tidal reach	Ammonia	1989-2002	1,128	-2.48	**0.014**
	Orthophosphate	1991-2013	1,21	-2.37	**0.027**
	Zinc	1989-2002	1,72	-1.84	0.070
	Lead	1989-2002	1,70	-0.08	0.934
2	DO	1973-2006	1,347	-2.38	**0.018**
Ouse Burn	BOD	1973-2006	1,380	-2.50	**0.013**
	Ammonia	1973-2006	1,370	-2.65	**0.008**
	Nitrate	1973-2006	1,359	-1.96	0.051
	Orthophosphate	1978-2006	1,259	2.25	**0.025**
	Zinc	1974-2006	1,347	-4.42	**<0.001**
	Lead	1976-2002	1,123	-5.24	**<0.001**
3	DO	1973-2019	1,494	4.03	**<0.001**
River Team	BOD	1973-2013	1,488	-10.59	**<0.001**
	Ammonia	1973-2019	1,536	-13.61	**<0.001**
	Nitrate	1973-2019	1,527	-0.47	0.639
	Phosphorus	1994-2019	1,276	-11.60	**<0.001**
	Zinc	1973-2019	1,517	-36.08	**<0.001**
	Lead	1973-2019	1,503	-21.89	**<0.001**
4	DO	1974-2020	1,564	2.18	**0.030**
Wylam Bridge	BOD	1974-2014	1,547	-5.16	**<0.001**
	Ammonia	1974-2020	1,582	-12.02	**<0.001**
	Nitrate	1974-2020	1,556	-2.94	**0.003**
	Phosphorus	1974-2020	1,441	-6.43	**<0.001**
	Zinc	1974-2020	1,480	-4.19	**<0.001**
	Lead	1974-2020	1,474	-11.11	**<0.001**

Table 2.2 Linear model summaries of changes in key water quality parameters in the Tyne catchment (S5-S8). Site numbers increase from downstream to upstream.

Site	Parameter	Periods	*df*	*t*	*P*
5	DO	1973-2019	1,457	0.41	0.684
Bywell	BOD	1973-2007	1,412	-3.58	**0.000**
	Ammonia	1973-2019	1,563	-4.68	**<0.001**
	Nitrate	1973-2019	1,455	0.91	0.362
	Orthophosphate	1974-2019	1,451	-3.87	**0.000**
	Zinc	1974-2019	1,454	-2.43	**0.016**
	Lead	1973-2019	1,203	-6.02	**<0.001**
6	DO	1974-2006	1,280	1.23	0.221
Wark	BOD	1974-2006	1,353	-0.86	0.391
	Ammonia	1974-2006	1,354	-1.01	0.312
	Nitrate	1974-2006	1,249	1.72	0.087
	Orthophosphate	1974-2006	1,231	0.74	0.458
	Zinc	1974-2006	1,273	-9.40	**<0.001**
	Lead	1974-1994	1,118	-13.25	**<0.001**
7	DO	1973-2020	1,540	-1.50	0.135
Warden Bridge	BOD	1973-2014	1,482	0.14	0.889
	Ammonia	1973-2020	1,557	-2.21	**0.028**
	Nitrate	1973-2020	1,517	-0.67	0.504
	Phosphorus	1974-2020	1,441	-5.78	**<0.001**
	Zinc	1974-2019	1,415	-1.81	0.070
	Lead	1974-2019	1,408	-4.88	**<0.001**
8	DO	1979-2020	1,415	0.80	0.427
Haltwhistle	BOD	1979-2007	1,338	-1.55	0.123
	Ammonia	1979-2020	1,436	-2.27	**0.024**
	Nitrate	1979-2020	1,339	-2.61	**0.009**
	Orthophosphate	1979-2020	1,243	-4.71	**<0.001**
	Zinc	1979-2013	1,347	-4.15	**<0.001**
	Lead	1979-2002	1,206	-4.79	**<0.001**

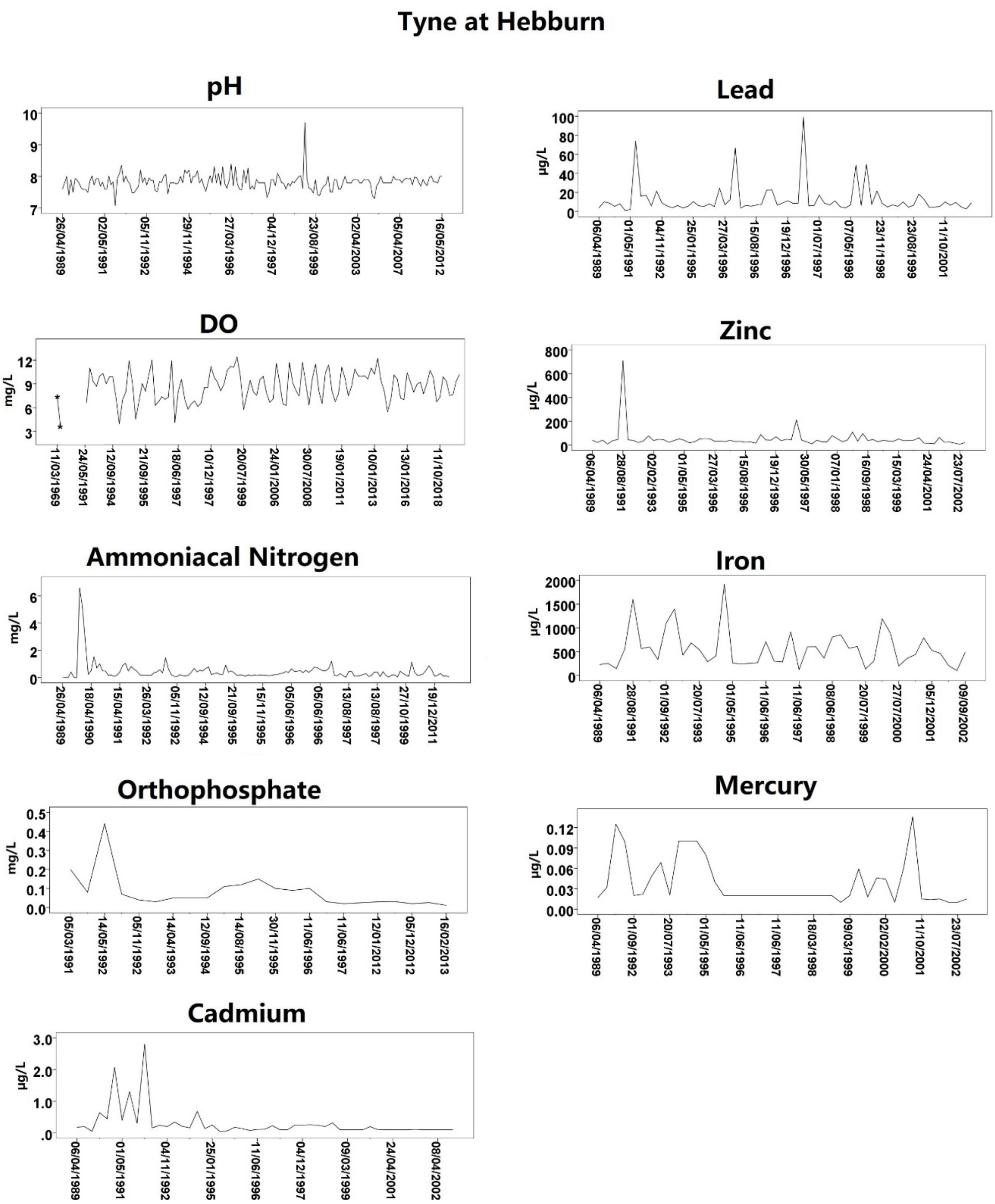


Figure 2.6 Key water quality parameters in the Tyne at Hebburn from 1989 to 2020. All metal element concentrations presented were 'total' values (samples not filtered). Note the different timescales on the panels. The DO data in 1969 and 1970 were extracted from Ratasuk (1972). Data between 2014 and 2020 were stitched from the Tyne at the Jarrow site.

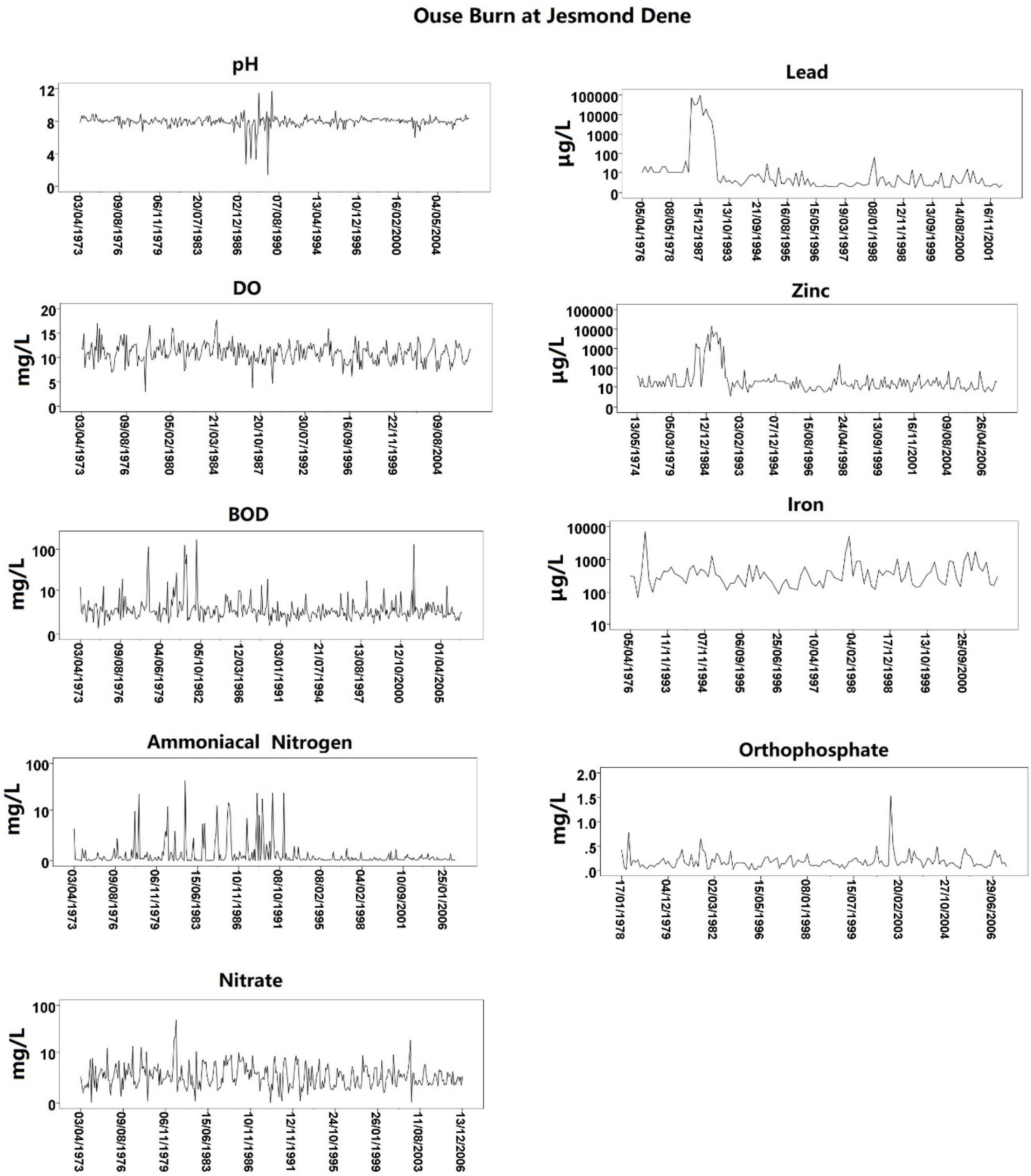


Figure 2.7 Key water quality parameters in the Ouse Burn at Jesmond Dene from 1973 to 2006. All metal element concentrations presented were 'total' values (samples not filtered). Notice: BOD, ammoniacal nitrogen, nitrate, lead, zinc and iron concentrations are on log scales. Note the different timescales on the panels.

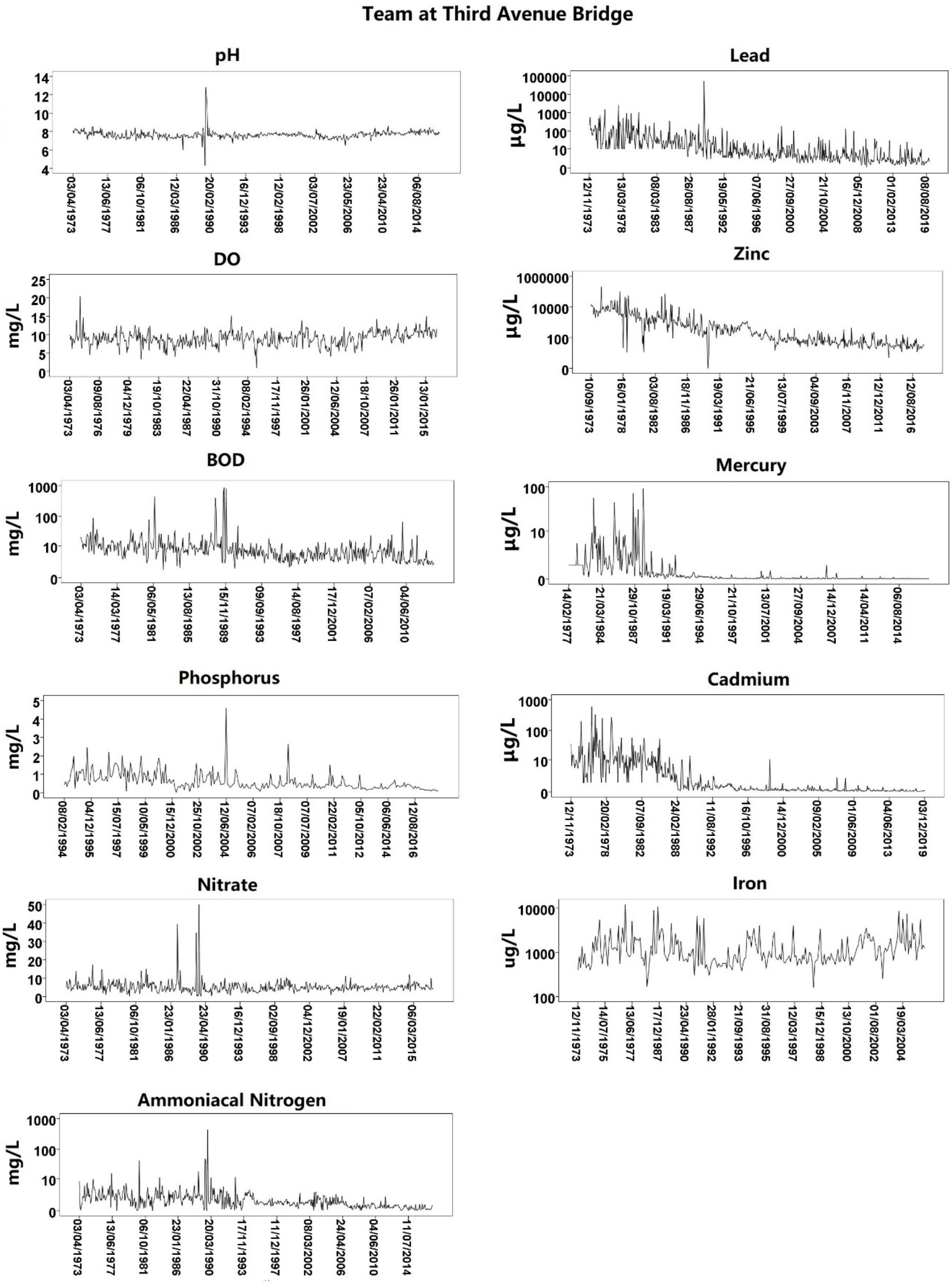


Figure 2.8 Key water quality parameters in the River Team at Third Avenue Bridge from 1973 to 2019. Notice: BOD, ammoniacal nitrogen, lead, zinc, mercury, cadmium and iron concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Phosphorus concentrations presented were total P.

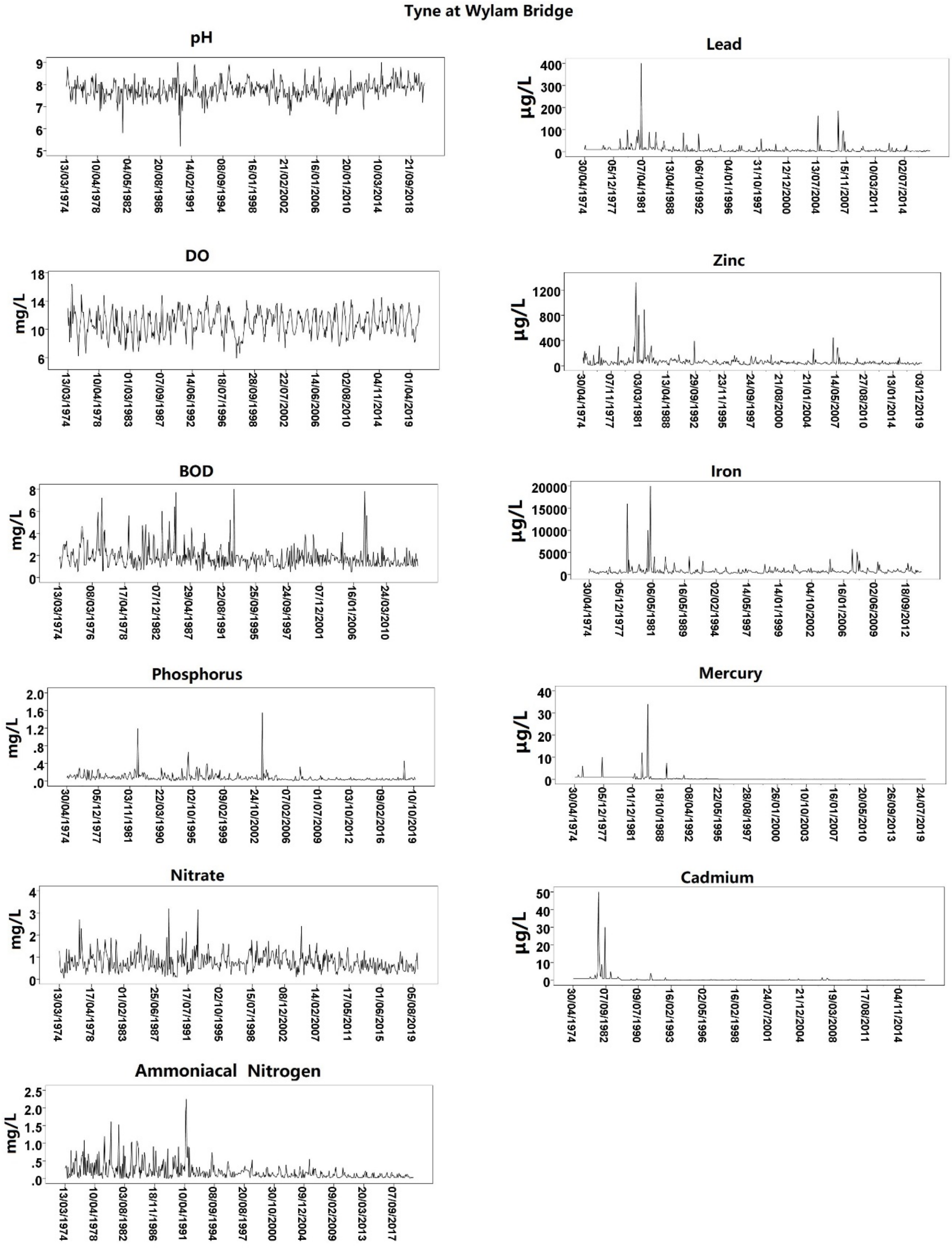


Figure 2.9 Key water quality parameters in the River Tyne at Wylam Bridge from 1974 to 2020. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Phosphorus concentrations presented were total P.

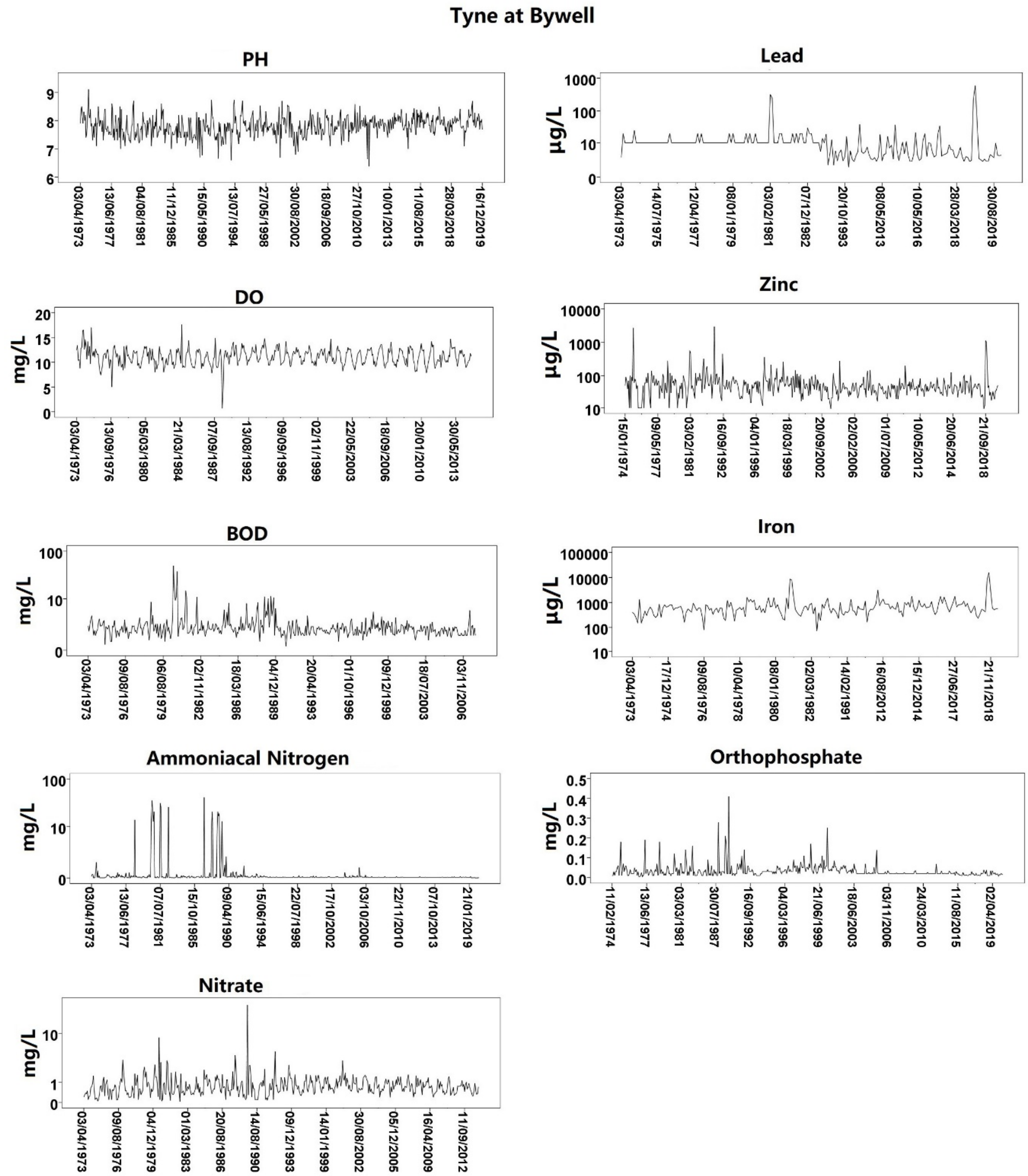


Figure 2.10 Key water quality parameters in the Tyne at Bywell from 1973 to 2019. Notice: BOD, ammoniacal nitrogen, nitrate, lead, zinc, and iron concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

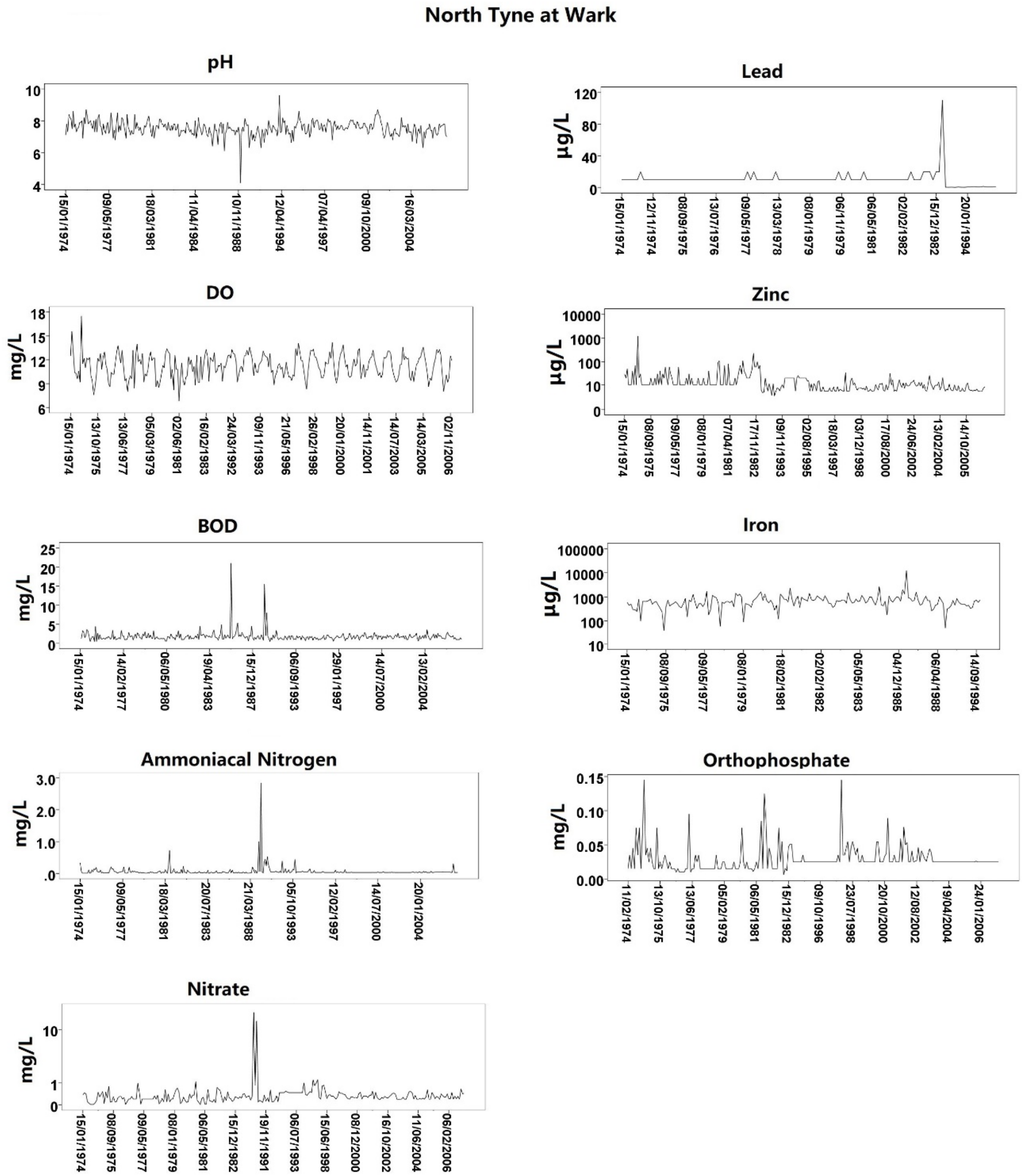


Figure 2.11 Key water quality parameters in the North Tyne at Wark from 1974 to 2006. Notice: nitrate, zinc, and iron concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

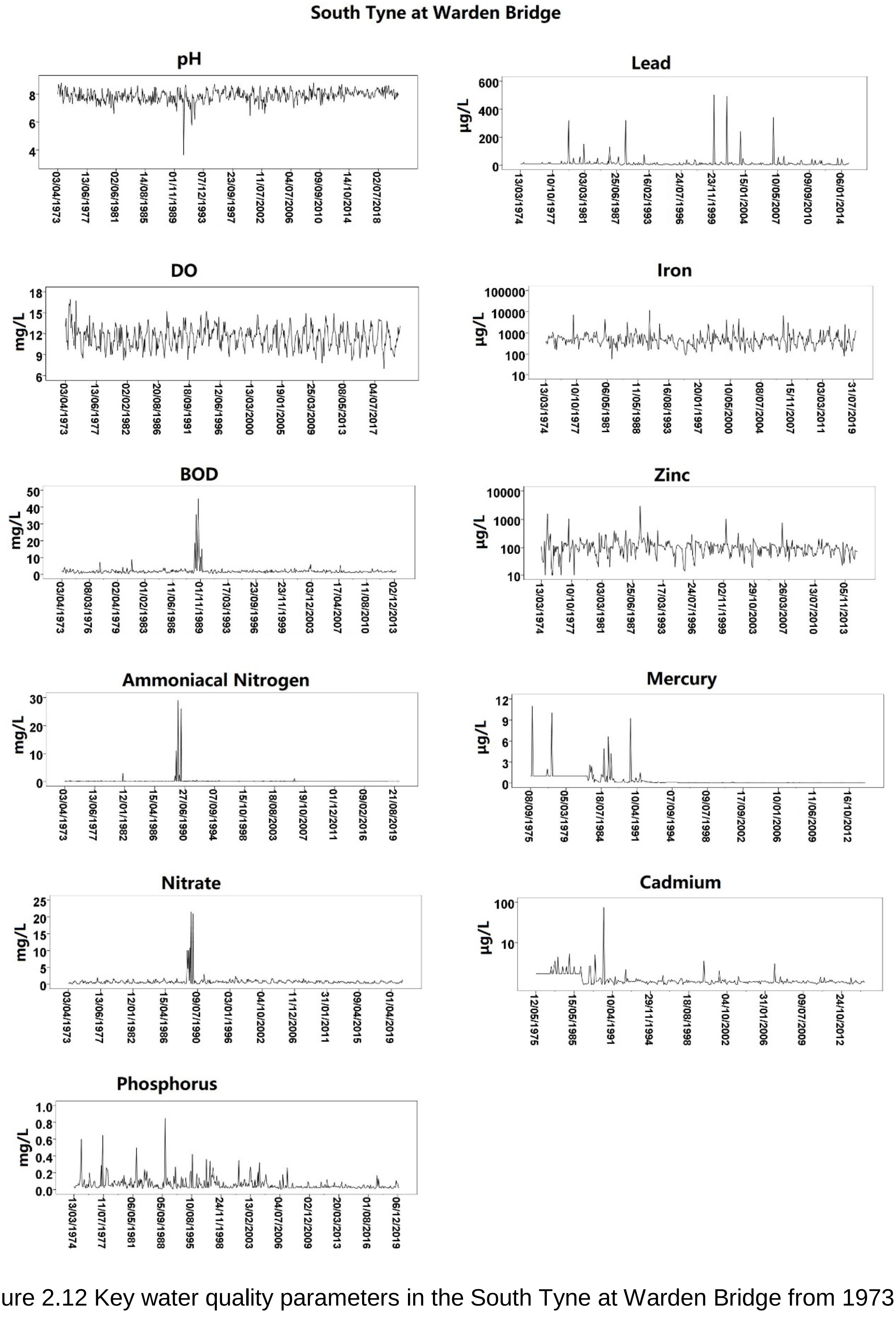


Figure 2.12 Key water quality parameters in the South Tyne at Warden Bridge from 1973 to 2020. All metal element concentrations presented were 'total' values (samples not filtered). Notice: iron, zinc and cadmium concentrations are on log scales. Phosphorus concentrations presented were total P.

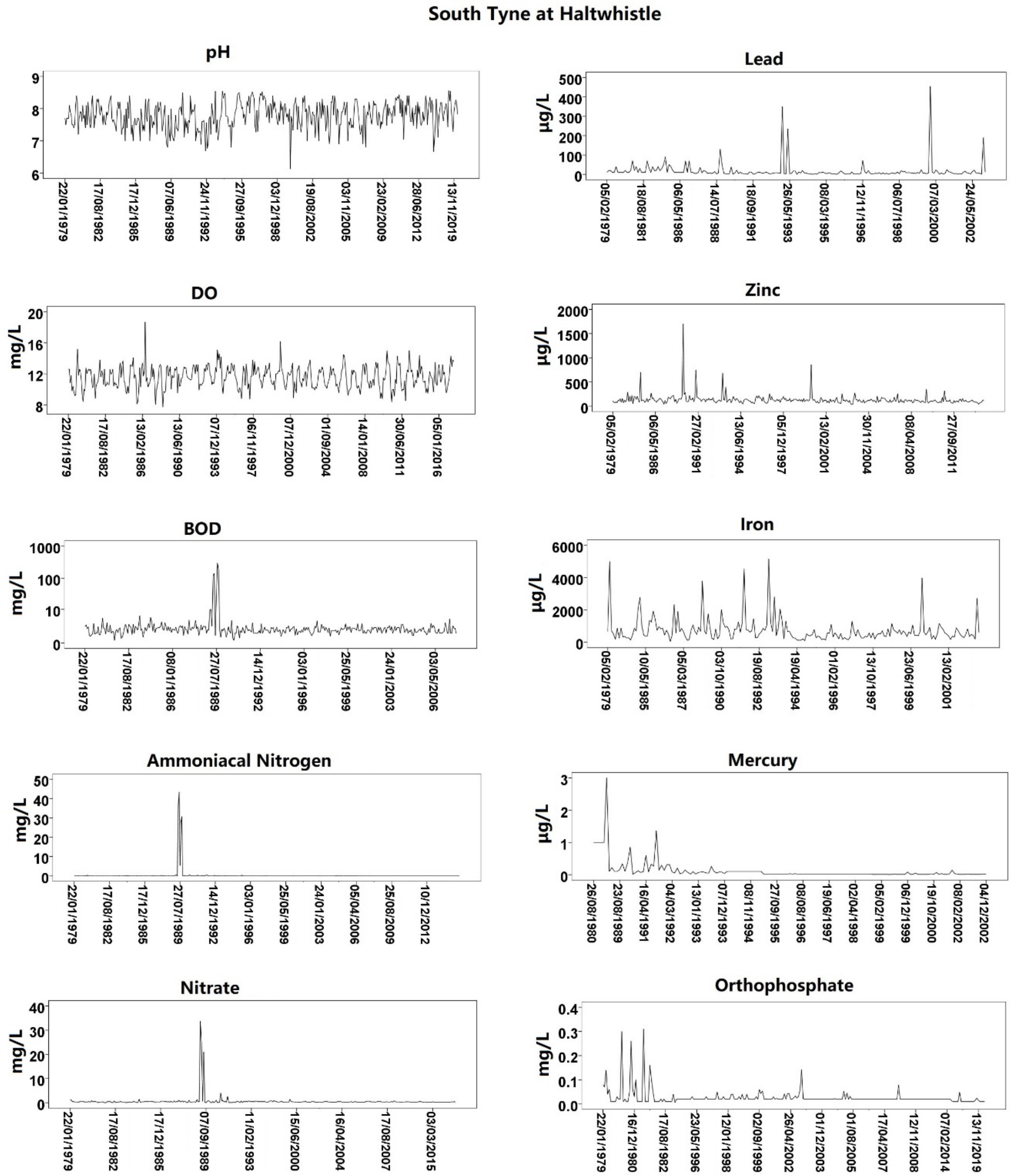


Figure 2.13 Key water quality parameters in the South Tyne at Haltwhistle from 1979 to 2020. All metal element concentrations presented were 'total' values (samples not filtered).

In 2015 and 2016, the majority of Tyne WFD water bodies were classified as ‘good’ for chemical status (Table 2.3). However, since 2016, new substances (e.g. perfluorooctane sulfonate) have been added to the assessment list and stricter standards have been developed for contaminants, which reflect the extent of these chemicals in the environment more accurately (Environment Agency, 2020d). The introduction of these has meant that no surface water bodies in England have met the criteria for achieving good chemical status in 2019 (Environment Agency, 2020d), and all water bodies in the Tyne catchment failed to achieve good chemical status in 2019.

The ecological condition in the Tyne slightly improved between 2015 and 2019, more water bodies shifted from poor to moderate status. However, more than half of water bodies [82/123 (66.7%)] still failed to reach good ecological condition in 2019, with the greatest pressures coming from hydromorphological modification and pollution from abandoned mines (Table 2.4). In addition, pollution from rural and urban areas, also contributed to pressures on water bodies.

Table 2.3 Ecological and chemical classification for surface waters in the Tyne catchment in 2015, 2016 and 2019 (cycle 2 of WFD).

Tyne catchment		Ecological status or potential					Chemical status	
Year	Number of water bodies	Bad	Poor	Moderate	Good	High	Fail	Good
2015	123	0	26	59	38	0	10	113
2016	123	0	24	63	36	0	14	109
2019	123	1	14	67	41	0	123	0

Table 2.4 Issues in the Tyne catchment preventing waters reaching good ecological status by cycle 2 (2015-2021) of the Water Framework Directive and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land management	Domestic General Public	Industry	Local and Central Government	Mining and quarrying	Urban and transport	Water Industry	Other	Sector under investigation	Total
Changes to the natural flow and levels of water	-	-	-	-	-	-	2	-	-	2
Pollution from rural areas	28	-	-	-	-	-	-	-	-	28
Pollution from abandoned mines	-	-	-	-	52	-	-	-	-	52
Pollution from waste water	-	-	-	-	-	-	9	-	-	9
Physical modifications	1	1	2	4	-	14	29	2	6	59
Pollution from towns, cities and transport	-	-	1	-	-	6	11	-	-	18
Non-native invasive species	-	-	-	-	-	-	-	-	-	0

2.3.1.5 Recovery of the Tyne fishes

With regard to the Tyne, only changes in the catches and other indicators of abundance of salmon and sea trout are considered here. Although recording of freshwater fish abundance in surveys across the catchment has been carried out by the EA and their predecessors these are not considered here, as no empirical work was carried out in this catchment for this thesis. By contrast, data on Tyne adult salmon and sea trout give a valuable indicator of post-industrial river recovery at a level of detail not available for the Wear or Tees. Along the northeast English coast to the north and south of the Tyne most fishing was by drift nets (gill nets at the surface) set offshore in coastal waters, and by T nets (which trap fish in an enclosure) set from the shore, the latter intercepting salmon and sea trout swimming in shallow water, often as they approach river mouths. The average annual drift net catch of salmon per licence on the Northumberland Coast showed a decreasing trend from the 1870s to the 1940s (Champion, 2003). Between 1951 and 1959, the annual combined net catch on the 'southern coast' (between Whitburn and Saltburn) was 3.6 for salmon and 779.6 for sea trout (annual data in Figure 2.14). During the same period, the annual combined net catch on the 'northern coast' (between Tweedmouth and Whitburn) was 5894.7 for salmon and 8060.8 for sea trout (annual data in Figure 2.14).

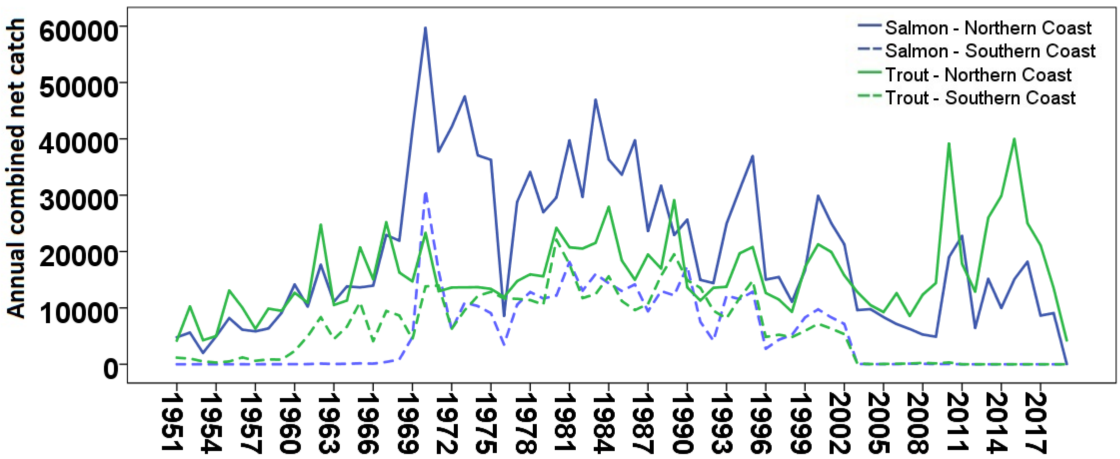


Figure 2.14 North East Northumbria Area, salmon and sea trout fishery annual net catch (all net types combined), from 1951 to 2019. Reasons for recent decline of catches on southern coast since 2003 was due to the closure of drift net fisheries. Figure generated using commercial catches reported to government using data sources described in Section 2.2.2.

The northern coast fishery started to recover in the 1950s and salmon capture per licence reached the pre-1870s level by the end of the 1960s when more effective nylon twine nets were largely used (Champion, 2003). Catches of sea trout on the southern coast started to recover in the early 1960s and salmon started to recover in the late 1960s. By 1970, monofilament nets were introduced and used by all licenced fisheries. This contributed to a dramatic increase in catch per net effort and the effort nearly doubled when compared with the previous netting method (Champion, 2003).

Because salmon and sea trout migrate along the Northumbrian coast before entering their home rivers (Potter and Swain, 1982), the Northumbrian and wider North East English net fishery, especially the drift nets, has been a ‘mixed stock’ fishery, exploiting multiple stocks from natal rivers. This has made effective management and conservation of salmon and, to a lesser degree, sea trout stocks, difficult and contentious. Stock is here defined as the proportion of a population considered to be a unit for fisheries management (Begg *et al*., 1999). Since Atlantic salmon and sea trout exhibit strong homing to natal rivers of origin, each major river is typically regarded as having its own stock(s). In order to reduce the mixed stock problem and thereby secure better salmon fishery management the Environment Agency has powers to make Net Limitation Orders under the Salmon and Freshwater Fisheries Act 1975. Net Limitation Orders are used to regulate salmon and sea trout net fisheries in England. Each order limits the number of licences for fishing with nets that may be issued in any specific fishery for up to 10 years. Net Limitation Orders were introduced in 1992 and 2002, and a permanent buyout of most drift net licences was achieved in 2003 (Environment Agency, 2008a). Since 2003, no licences were issued to the Southern Northumbrian coast drift net fishery, both salmon and sea trout catches from that source were reduced to zero. In December 2018, national byelaws closed the drift net fishery completely (Environment Agency, 2020e). The beach net fishery, comprising T nets and J nets (J nets, similar to T nets, are used from Yorkshire shores) was closed for salmon as a conservation measure, but on the NE coast fishermen were allowed to continue to fish for sea trout only and allocated larger sea trout quotas as a compensatory measure (Environment Agency, 2020e). Experiments are ongoing to minimize salmon capture in modified T-nets (Environment Agency, 2020d)

In 1934, the Tyne river net fishery (seine nets and some T nets) was banned by byelaw to help restore the rod fishery (McParlin, 2011). Following the improvement of water quality, the salmon population in the Tyne started to recover during the 1960s, and the total rod catch showed a dramatic increase from the late 1980s (Figure 2.15). The total rod catch per year increased from three in 1960 to 5115 in 2010 (Figure 2.15). The sea trout population showed a similar trend in recovery compared with salmon, the rod catch per year increased from 15 in 1960 to 2687 in 2010 (Figure 2.15).

The rod catch per unit effort (catch per licence day) of salmon in the Tyne steadily increased between 1993 and 2004, and then fluctuated between 2004 and 2017, with a peak in 2011 (Figure 2.16). The rod catch per unit effort of sea trout showed an increasing trend between 1993 and 1998, and fluctuated between 1999 and 2017.

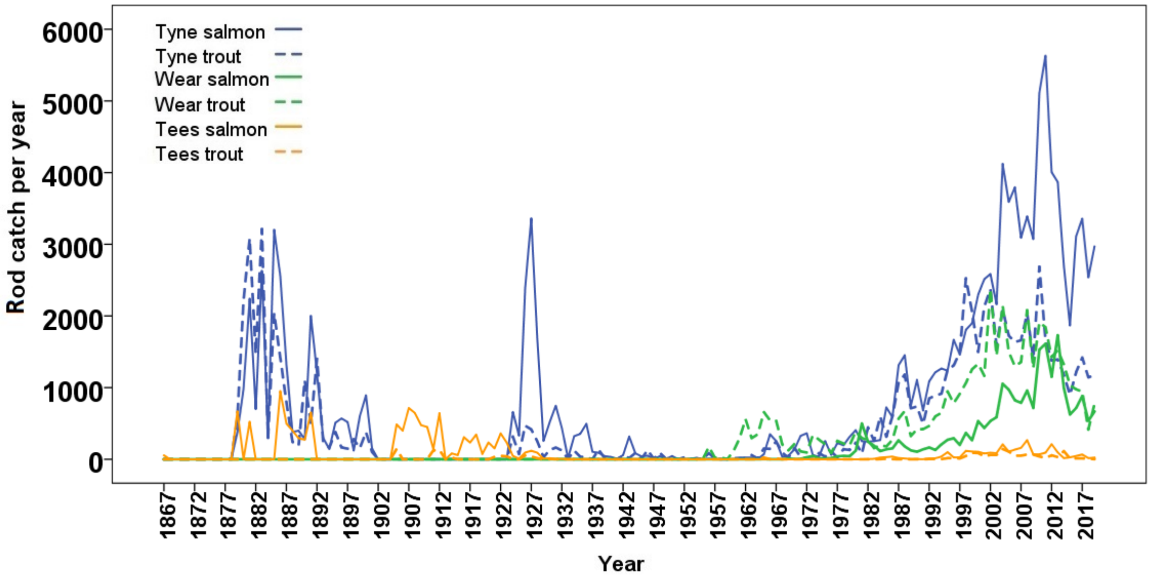


Figure 2.15 Annual declared rod catch of salmon and sea trout in the Tyne, Wear and Tees from 1867 to 2019. Note that before 1950 rod catch data suffer from gaps in historic and river specific recording and can serve only as a crude indication of temporal patterns of catch.

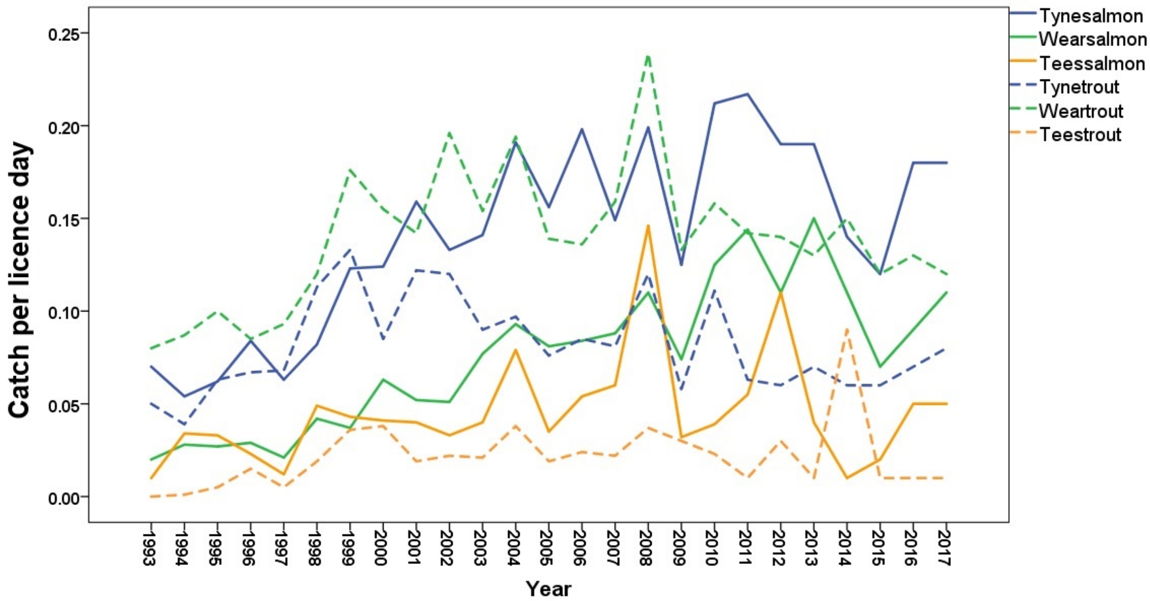


Figure 2.16 Catch per licence day of salmon and sea trout in the Tyne, Wear and Tees from 1993 to 2017. CPUE for 2018 and 2019 are not given as effort was not available in the respective Environment Agency reports.

At the time of the Kielder Dam plan, in order to help recover the salmon population in the

North Tyne, Northumbrian River Authority proposed a plan of building a fish pass on the Kielder dam to provide upstream access. However, subsequent habitat surveys showed that the upstream major tributary could not provide sufficient spawning habitat to salmon due to previous intensive gravel extraction. After the fish pass construction plan was discarded, an alternative plan was proposed (and stipulated by legislation in parliament): build a hatchery and restock the river with salmon parr/eggs to compensate for the loss of salmon production caused by the construction of the reservoir (Cave, 1985). The Kielder hatchery stocking programme began in 1979 and continues today. During the spawning season, electro-netting is conducted below the main dam, eggs and milt are stripped from the mature salmon, then sent back to the hatchery for hatching. After stripping, the adult salmon are released back to the river. The legal mitigation agreement is to stock a total of 160,000 0+ and/or 1+ juvenile salmon in the Tyne catchment annually (Milner *et al.*, 2004).

Before the construction of Kielder hatchery, between 1954 and 1978, limited stocking of salmon into the Tyne system occurred, comprising salmon eggs, parr and smolts, derived from unspecified Scottish sources. Between 1965 and 1977, eggs were planted at rates of 25,000 to 150,000 annually (Milner *et al.*, 2004). Between 1989 and 2017, an average of 379,898 (range, 46,000 to 638,898) comprised fry (range, 0 to 400,000), parr (0+ range, 0 to 572,799; 1+ range, 0 to 140,214) and smolts (range, 0 to 5,319) were stocked in the Tyne catchment (Figure 2.17). Between 1983 and 2000, batches of 1+ salmon parr reared at Kielder hatchery were marked with coded wire microtags (CWTs) (Jowitt and Russell, 1994). Tag returns were recorded from both the Tyne rod fishery and net fishery that operates along the North East coast and around the Tyne mouth. Estimates of the long term (1980-2000) weighted mean contributions to the North East Coast Fishery and the Tyne rod catch were 1.5% (range 1.2-2.0%) and 6% (range 3-14%) (Milner *et al.*, 2004). The stocking in the Tyne continues to the present day, and from 1989 to 2017, a total of 10.9 million salmon were released in the Tyne catchment (Figure 2.17).

In 1996, a resistivity fish counter started operation at Riding Mill. The counter relies on the principle that fish have a lower electrical resistance than the water, so when fish (and other animals, or even some other types of object) pass the fish counter, the counter detects an increase in conductivity between them, indicating passage of a fish/animal (Van Der Waal, 2014). Most resistivity fish counters are of sufficient sensitivity to be suited to recording anadromous salmonids (>~40 cm) but cannot distinguish between species and only since 2003 was video capture added to the Riding Mill counter to help distinguish between salmon and trout records. These data have been subject to intensive analyses (Van Der Waal, 2014) to which the reader is directed for more information. Total recorded fish counts (salmon plus large trout, primarily sea trout) since 1996 have ranged from 15,219 (2000) to 48,668 (2004) (Figure 2.18).

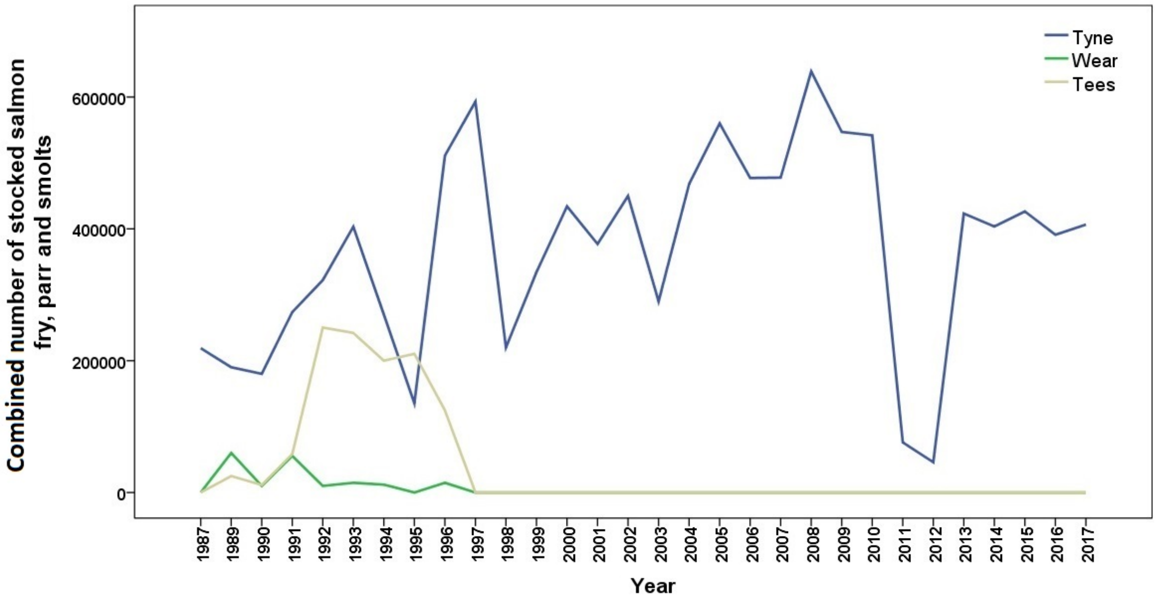


Figure 2.17 Annual variation of salmon stocking in the Tyne, Wear and Tees from 1987 to 2017. Notice: the 1988 stock data is currently not available and the 2018 and 2019 reports do not provide stocking data.

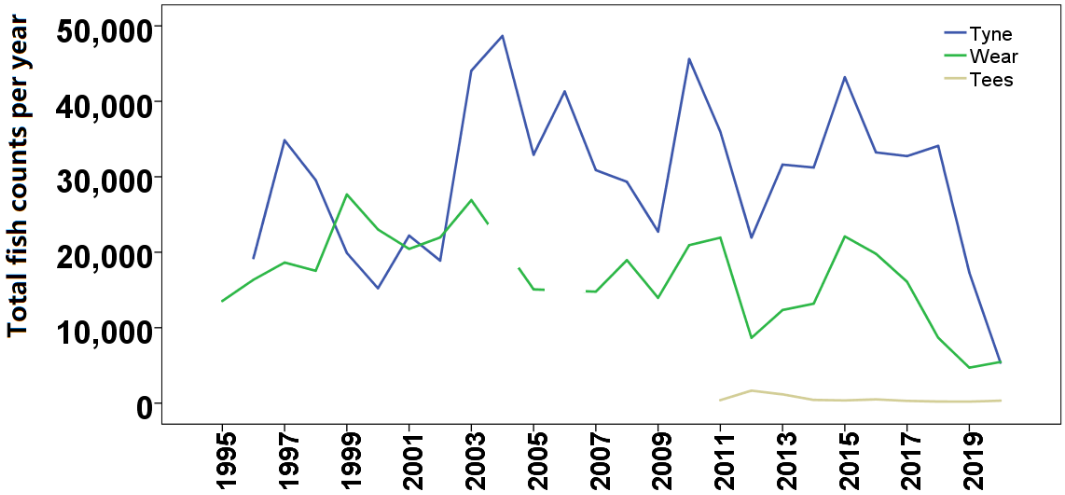


Figure 2.18 Annual upstream counts of migratory adult salmonids (salmon plus sea trout) from 1995 to 2020 on the Tyne (Riding Mill), Wear (from February 2015 onwards these represent a combined count from both Framwellgate and Freemans counters, Durham) and Tees (Tees Barrage). Note: the Wear fish counter did not operate during the salmonid upstream migration periods in 2004 and 2006. The Tyne fish count data from April to July 2020 were unavailable so 2020 data are a minimum.

Not all salmon and sea trout are recorded by the Riding Mill counter, as fish can ascend the weir directly at high flows; the EA assumes 10% bypass the counter (P. Rippon pers. comm.). River Tyne adult salmonids mainly migrate between June and November, monthly counts peak in October, though they differ between the species. Daily peaks in counts closely correlate to the occurrence of river flow peaks (Environment Agency, 2008c).

2.3.2 River Wear

2.3.2.1 History of Wear salmon

From mediaeval times to the Industrial Revolution, the River Wear afforded an abundant supply of Atlantic salmon. Between 1348 and 1437, monks of Finchale Monastery, derived a considerable revenue by selling salmon and trout (Commissioners for the British fisheries, 1861). The Convent of Durham bought hundreds of salt salmon each year between 1530 and 1536 (Commissioners for the British fisheries, 1861). For a long period, it was a condition inserted in all indentures of apprenticeship that the apprentice was not to be compelled to eat salmon more than three days a week (Anonymous, 1861). The main spawning areas for salmon in the Wear are located between Bishop Auckland and Wearhead (Environment Agency, 2008a); whether this was the case hundreds of years ago is unclear.

2.3.2.2 Connectivity and habitat deterioration in the Wear

Although the River Wear supports a migratory salmonid fishery, the access to many of the Wear tributaries is restricted due to the presence of impassable natural and man-made obstructions (e.g. waterfalls, weirs) (Environment Agency, 2008a). The most famous historic barrier to upstream fish migration in the Wear is suggested to be Lumley Lock. In the beginning of the 19th Century, a dam (Lumley Lock) was constructed at the tidal limit across the Wear, near Lumley Castle, following the order of predecessors of Lord Scarborough. Before then, no barrier existed to the passage of salmonids upstream towards the main spawning grounds in Weardale. Successful migration and spawning by anadromous salmonids in the Wear was prevented by Lumley Lock (Anonymous, 1861). Although most of Lumley Lock was washed away by a flood in winter 1820, salmon remained absent from the upper part of the river (Commissioners for the British fisheries, 1861). For several years no salmon were observed upstream of Lumley Lock and only a few were observed below it. Lumley Lock was destroyed by floods in January 1854, and immediately after it was removed, sea trout (often referred as white trout or bull trout in early angling literature) ascended and their population increased in the upstream reach (Anonymous, 1863). However, salmon was still absent from the upstream reach until an individual was caught in the Durham reach in 1866 (Anonymous, 1866).

Apart from Lumley Lock, several more weirs were recorded to obstruct fish migration in the River Wear. A weir in the main Wear located about one mile (1.6 km) upstream of Wolsingham was observed to block upstream fish migration in 1860 (Commissioners for the British fisheries, 1861). A dam near Bishop Auckland was observed to block upstream salmon migration (Anonymous, 1877). In the River Browney, two dams constructed four miles (6.4 km) downstream of its source completely blocked fish from ascending upstream to spawning habitat (Commissioners for the British fisheries, 1861).

Besides instream barriers, the Wear also suffered from other anthropogenic influences. Like the Tyne, commercial gravel extraction occurred in the middle Wear catchment between 1945 and 1968 (Wishart *et al.*, 2008). The impacts of gravel extraction on river habitat at two previous working sites was studied by Wishart *et al* (2008). At Wolsingham, extraction work included unsystematic pit excavation in the riverbed, with regular rerouting of flow to expose new areas of the riverbed to be worked and pre-extracted pits refilled with sediment (Wishart *et al.*, 2008). An average of 36,000 tonnes gravel was extracted each year at the central portion of the site. At Harperley Park, dragline excavation occurred between 1960 and 1968, and an average of 100,000 tonnes gravel was extracted per year (Wishart *et al.*, 2008). Commercial river channel gravel extraction in Weardale ended in 1968, and following a brief period of floodplain extraction along the lower section of the Harperley Park reach, gravel extraction ceased entirely in the early 1970s. Gravel extraction involved the removal of a large volume of sediment from channel bars and the riverbed in the River Wear and significantly reconstructed the river channel from braided to single thread planform (Wishart *et al.*, 2008). Channel modification and agricultural practices, as well as disused mine pumping practices in the Wear catchment, has led to the reduction in summer flow and increase in winter spates over recent decades, which has the potential to influence salmon migration and survival following hatching (Environment Agency, 2008a).

2.3.2.3 Pollution of the Wear

Apart from barriers, pollution was also considered to be another major problem for fish in the Wear and its tributaries. Like the south Tyne catchment, coal mining in the Wear catchment began in the Late Middle Ages. Coal production reached a peak in 1913 and in 1923 there were 170,000 miners working in County Durham. The middle and lower reaches of the Wear were heavily affected by the coal mining history. It was reported by the Ministry of Agriculture and Fisheries Standing Committee that the middle reach of the river carried high levels of coal tar acids in September 1928 (Longwell and Roberts, 1929). By the mid-20th Century, nearly 20-30 million tons of coal was extracted from the Durham coal mines on a yearly basis (McParlin, 2011), and the river still suffered significant pollution from coal mining, coke works and other effluents in the 1950s (Whitton *et al.*, 1998). Coal mining required substantial pumping, to maintain the water level in the working area and protect mines from flooding, this action discharged large volumes of polluted water into the river system. The status of the River Wear was classified as “ruined by mines” in the late19th Century and fish preservation in County Durham had been given up entirely (House of Commons, 1876).

Metal mining in the North Pennines started during Roman times, increased rapidly during the 19th Century, and it was estimated there were more than 100 mines in operation in the early 20th Century (Wishart *et al.*, 2008). In the upper Wear catchment within the North Pennine Orefield, mineral veins were rich with hydrothermal Pb-Zn-F-Ba mineralisation and were exploited by hydraulic, surface and underground mining methods (Lord and Morgan, 2003). Besides mining, associated mineral processing and smelting activity also existed in this area. Some tributaries of the Wear (e.g. Bollihope Burn and River Browney) were used as conduits for taking off the foul water which is pumped out of mines (Commissioners for the British fisheries, 1861; Anonymous, 1867). The former signs of “hushing” can still be found in the landscape nowadays. Followed by the decline of industry after the Second World War, many pits closed in the 1950s and 1960s in County Durham. The last colliery in the Durham coalfield closed in 1994. Limestone was, and still is, extensively quarried for construction aggregate and cement manufacture usage (Lord and Morgan, 2003).

In the upper Wear catchment, metal mining had a long lasting effect on river sediment. A study in 1999 indicated that severe contamination with lead exists throughout the entire upper catchment river network including both main river and tributaries, and persists in sediments of the main river (Lord and Morgan, 2003). Severe zinc contamination was mainly concentrated in the main River Wear and concentrations declined downstream towards Wolsingham (Lord and Morgan, 2003). Arsenic contamination was mostly concentrated through the Rookhope Burn and from Rookhope Burn-Wear confluence to beyond Stanhope as well as in the lower Bollihope Burn. Arsenic contamination appeared to decrease in the downstream reach of the upper catchment (Lord and Morgan, 2003). Other metal contaminants such as barium and iron were present in lower concentrations in both main River Wear and its tributaries in the upper catchment (Lord and Morgan, 2003).

In the lower Wear catchment, urban and industrial pollution were the main forms of pollution in the freshwater reach. In the late 1800s and early 1900s, as in the Tyne, hardly any houses or factories controlled their pollution discharges and most of these went direct to the river, or were able to leach to the river from midden heaps. The increased population density associated with mining and heavy industry caused intensive pollution, parallelling the pattern on the Tyne, but including much greater pollution on tributaries from the numerous pit villages. Even from the 1950s until recent decades, the river in the middle and lower reaches was significantly affacted by sewage treatment works discharges (Environment Agency, 2008a). The Wear catchment had a human population of almost 500,000 in 2008, and major population was concentrated in the eastern half of the area including the two cities of Durham and Sunderland (Environment Agency, 2008a).

In the lower Wear catchment, a large paper mill was constructed in the River Browney near the Browney-Wear confluence and some paper works were located two or three miles (~4 km) further upstream. Discharges from these works flowed into the river and caused significant pollution (Commissioners for the British fisheries, 1861). Large cokeworks were also present on the lower Browney in the area of Langley Park. The sewage water of Durham and Bishop Auckland entered the river directly; the gasworks emptied their waste into the river in Durham and Bishop Auckland and dye works in Durham also discharged into the river (Commissioners for the British fisheries, 1861). The pollution of the Wear, caused by the hushing from the metal mines in the higher reaches of the Wear and the wastewater from the collieries and ironworks near Durham are recorded to have rapidly diminished the breeding of fish in the river and almost put an end to the sport of angling (Anonymous, 1873). Within the estuary area, the city of Sunderland had a massive industrial expansion since the 18th Century. By the mid-1700s, Sunderland became one of the largest ship builders in England (Short and Tetlow, 2012). During the 19th Century, ship making greatly developed due to the need for ships to carry coal to London (McLean, 1995). From then until the late 19th Century, Sunderland was the largest shipbuilding centre in the world, making approximately one-third of the UK’s ships (Short and Tetlow, 2012). Other important industries in Sunderland included glass, pottery, rope making, and coal exportation. The shipbuilding works led to a heavily modified estuary environment, including bank reinforcement and river deepening. Along with the rising population in Sunderland, the disposal of untreated domestic and industrial waste water became another pollution issue for the Wear estuary.

2.3.2.4 Recovery of the Wear habitat

In June 1854, a meeting was held in Durham to consider the best methods of increasing the stock of salmonids in the River Wear. The meeting resolved to cooperate to propagate salmon by artificial means for stocking (Anonymous, 1854). A meeting was held in October 1864 in Bishop Auckland, to agree how to increase fish spawning in the Wear and its tributaries, as well as protect the fish from coal mining pollution (Anonymous, 1864). In 1873, meetings were held to try to solve industrial pollution in the Wear, plan a fish pass on a dam near Bishop Auckland and to try to establish a Fishery Board for the Wear (Anonymous, 1873, 1877).

Since 1980, following the closure of coal and metal mines, and associated activities, particularly coke making, the freshwater quality in the River Wear improved significantly (Milner *et al.*, 2004). Historically, abandoned coal mines near the Wear have been dewatered and the discharges drained into the middle and lower reaches directly which has potential negative impacts on salmon. More than £140K cost was invested by the NRA to investigate mine waters and reduce deep coal mine pumping in 1995 (National Rivers Authority, 1995d). By the late 2000s, most of these discharges had been relocated to the coast and no longer affected flow and water quality in the main river (Environment Agency, 2008a).

By 1995, there were approximately 143 Sewage Treatment Works (STW) operating in the Wear catchment. Northumbrian Water Ltd (NWL) own 83 of these works, of which 50 serve populations in excess of 250 (National Rivers Authority, 1995d). The remaining STWs serve small private developments and were operated by the owners. A number of STWs were causing deleterious impacts on the receiving watercourses (National Rivers Authority, 1995d), including Sedgeletch STW (Herrington Burn), Crookhall STW (Stockerley Burn) and Hustledown STW (Twizell Bum) (Environment Agency, 1997c). Because some of the water quality problems were due to the effects of discharges from NWL STWs or combined sewerage overflows (CSOs) to the watercourse, NWL was requested to upgrade some of their STWs and CSOs (Environment Agency, 1997c). In order to protect and improve the water quality of the Wear estuary, NWL was requested to intercept the crude sewage discharges to the Wear Estuary and divert to Hendon Sewerage Treatment Works (Environment Agency, 1997c). In addition, NWL was requested to install secondary sewage treatment at Hendon STW (Wear estuary) by 2001 (Environment Agency, 1997c).

The EA and Natural England also sought to reduce diffuse pollution (e.g. from agriculture, urban areas and roads) and take action on the discharges and abstractions made by companies (e.g. Northumbrian Water) (Environment Agency, 2008a). The EA proposed nutrient removal from STW effluent in the Northumbria Area to improve 68 km of river (Environment Agency, 2008a). The overall improvements to sewage treatment have led to significant improvements in water quality of the Wear (Milner *et al.*, 2004). Under the old General Quality Assessment scheme, by 2008, the main river Wear and majority tributaries achieved good water quality (Environment Agency, 2008a). However, GQA was an unrealistically generous classification system, failing to take account of many factors impacting the ecological health of river systems since for example Figure 2.19 shows good water quality for the River Deerness, but subsequently this repeatedly failed WFD quality tests.

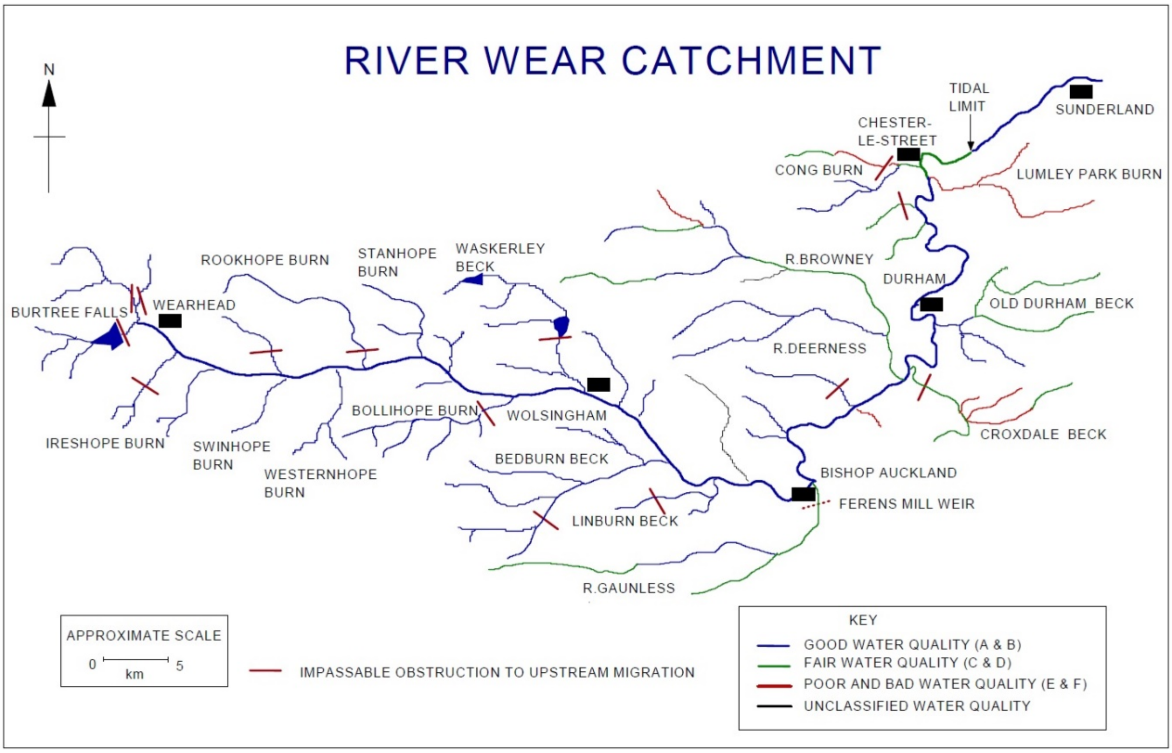


Figure. 2.19 Water quality of major tributaries and obstructions to fish passage in the Wear catchment by 2008. Source: Environment Agency (2008a).

In the Wear catchment, 12 sites were chosen to examine the long-term chemical trends (Figure 2.20). In order to obtain a more continuous data run from the 1980s to 2020 within the tidal reach, data from three adjacent sites were stitched. With regard to other sites chosen, the Lambton Bridge site, located at the tidal limit, reflects the change of chemical trends in the lower main river. The Wear at Stanhope is located in the upper main river, and reflects the chemical change caused by intensive metal mining works in the upper reaches. One site located in Old Durham Beck, represents a tributary which was historically badly polluted. Two sites were located in the Cong Burn sub-catchment, one site located in the River Deerness, one site located in the Brancepeth Beck, and one site located in the Bedburn Beck. These four sub-catchments are study streams in which empirical work was undertaken in Chapter 5. Coal mining activities of urban development occurred along the former three streams. One further site was located in the Rookhope Burn, which was historically polluted, and this would be indicative of the metal mine rich areas of the Wear.

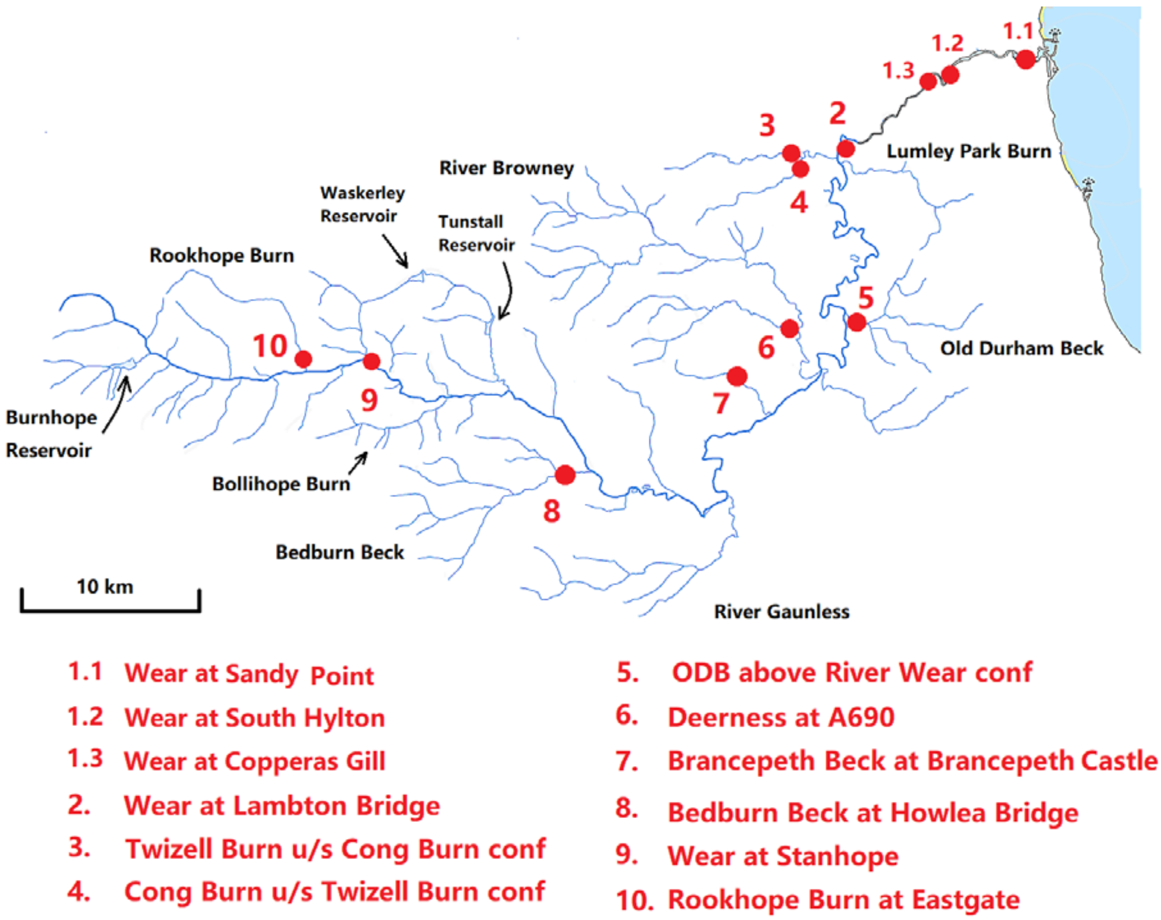


Figure 2.20 Wear catchment and sites with long-term chemical trends for which data are presented in this thesis. 'Conf' refers to confluence and 'u/s' refers to upstream.

Regrettably, data for the Wear estuary only ran from 1988 to 2020, and no metal elements were measured after 2002. The pH value varied considerably between 1989 and 1998, then slightly declined and fluctuated less (Figure 2.21). Of the determinands analysed statistically, DO increased and ammonia and orthophosphate decreased significantly (Table 2.5). The DO concentration varied considerably from ~6 mg/L to ~12 mg/L between 1988 and 2010, then slightly increased and fluctuated less after that (Figure 2.21). Ammoniacal nitrogen largely declined between 1988 and 1990, then became stable around 0.5 mg/L (Figure 2.21). For iron, lead, zinc and mercury concentrations, a few peaks were recorded in 1989 and the early 1990s, then both declined and fluctuated less, although two peaks of mercury were detected in 2001. These trends are indicative of an improvement in water quality associated with reduced oxygen-demanding and industrial waste in the estuary.

At Lambton Bridge, the tidal limit, (1973-2018), BOD, ammoniacal nitrogen, phosphorus, lead, zinc, decreased significantly and DO and nitrate increase significantly (Table 2.5; Figure 2.22). For nitrate, multiple high values around 30 mg/L occurred in 1990, then it dramatically reduced to around 4 mg/L and became stable. Total phosphorus concentration varied between 0 and 2 mg/L before 2006, then reduced to around 0.1 mg/L by 2019 (Figure 2.22). Other chemical components did not show any clear trends. Periodic elevated values of substances such as cyanide were evident in the 1990s (Figure 2.22).

At the Twizell Burn site (1979-2019) DO, and nitrate increased significantly while BOD, ammonia, orthophosphate and zinc decreased significantly (Table 2.5). Orthophosphate strongly declined between 1995 and 2006, then it became stable between 2006 and 2019 (Figure 2.23), reflecting improved sewage treatment upstream (Hustledown STW). Ammonia decreased slightly and DO increased slightly over the same period (Figure 2.23). At the Cong Burn site BOD, ammonia, nitrate and orthophosphate decreased over the analysis periods (Table 2.5). Ammonia and BOD were reduced after 1993, with fewer and smaller peaks, probably reflecting improved water treatment (Figure 2.24). Other chemical components did not change markedly during sampling periods.

In Old Durham Beck (1973-2019), BOD, ammonia, zinc and lead decreased significantly and nitrate increased significantly (Table 2.5). Ammonia and BOD declined slightly since 1990, although quite frequent peaks remained after 2000 (Figure 2.25). Although peaks in nitrate reduced after 1990, baseline nitrate levels have increased since the mid-1990s (Figure 2.25), probably reflecting the intensive arable farming within the catchment. This is also reflected in the continued high but variable phosphate levels. Periodic high iron levels were recorded, potentially linked to the coal mining history of the area. No clear trends were observed in other chemical components during sampling periods.

At the Deerness site (1973-2019), BOD, ammonia, zinc and lead decreased significantly and nitrate increased significantly over the data timescales (Table 2.6). The concentration of ammoniacal nitrogen declined markedly since 1992, and it became stable between 1992 and 2019 (Figure 2.26). At the Brancepeth site (1991-2004), the concentration of ammoniacal nitrogen declined significantly, other chemical parameters were relatively stable between 1984 and 2014 (Figure 2.27; Table 2.6).

At the Bedburn Beck site (1991-2019), pH values varied considerably between 1991 and 2007, then slightly increased and became more stable after 2007, possibly linked to changes in forestry practices, since commercial coniferous forestry tends to promote acid flushes (Figure 2.28). Of those determinands analysed statistically over the periods of data availability, ammonia, nitrate, orthophosphate and zinc all decreased significantly (Table 2.6). Ammoniacal nitrogen concentrations were slightly varied between 1991 and

1993, then reduced and became relatively stable after that (Figure 2.28). It is notable that nitrate levels were lower than sites further downstream and probably reflects the dominance of forestry and lack of intensive farming locally. Zinc concentrations in Bedburn Beck were similar with the Twizell Burn, which were slightly higher compared with Cong Burn, Deerness and Brancepeth Beck.

At the Stanhope site (1973-2019), ammonia, nitrate, zinc and lead levels decreased significantly over the periods for which data were available (Table 2.6). Ammoniacal nitrogen, nitrate, and zinc concentrations varied during the 1980s, then became relatively stable and lower after that (Figure 2.29). Cadmium concentrations reduced from 20 ug/L in 1978 to 0.2 ug/L in 1987, and became stable. Peaks in lead and zinc probably relate to resuspension due to high flows; the incomplete time series makes it difficult to tell the true degree to which these metal levels have reduced and stabilized, but wherever channel and riparian sediments are reworked during spates it is likely these peaks will be generated. No clear trends were observed in other chemical components.

In Rookhope Burn (1976-2019), oxygen increased significantly and ammonia, nitrate, orthophosphate, zinc and lead all decreased over time (Table 2.6). Ammonia varied substantially between 1979 and 1994, then slightly declined and became relatively stable (Figure 2.30). High concentrations of lead, iron and zinc were recorded between 1978 and 1991, but then declined (Figure 2.30). Although another high peak of zinc concentration (linked to an adit outflow event) was detected in early 2000, this steadily decreased to the previous level by 2002. Cadmium concentrations reduced from 10 μg/L in 1981 to less than 1 μg/L after 1991, and became stable.

Table 2.5 Linear model summaries of changes in key water quality parameters in the Wear catchment (S1-S5). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea.

Site	Parameter	Periods	*df*	*t*	*P*
1 Tidal reach	DO	1989-2020	1,62	2.64	**0.011**
	Ammonia	1988-2002	1,115	-3.54	**<0.001**
	Orthophosphate	1989-2020	1,27	-3.09	**0.005**
	Zinc	1989-2002	1,45	-1.71	0.094
	Lead	1989-2002	1,43	0.03	0.974
2 Lambton Bridge	DO	1973-2019	1,515	3.13	**0.002**
	BOD	1973-2014	1,539	-8.35	**<0.001**
	Ammonia	1973-2019	1,591	-11.81	**<0.001**
	Nitrate	1973-2019	1,537	2.00	**0.046**
	Phosphorus	1974-2019	1,389	-11.85	**<0.001**
	Zinc	1974-2019	1,427	-5.84	**<0.001**
	Lead	1974-2019	1,426	-5.62	**<0.001**
3 Twizell Burn	DO	1979-2019	1,270	4.15	**<0.001**
	BOD	1979-2006	1,81	-1.98	**0.050**
	Ammonia	1979-2019	1,287	-4.12	**<0.001**
	Nitrate	1979-2019	1,283	2.98	**0.003**
	Orthophosphate	1995-2019	1,232	-19.49	**<0.001**
	Zinc	1989-2011	1,108	-2.21	**0.029**
4 Cong Burn	DO	1979-2019	1,205	1.13	0.262
	BOD	1979-2006	1,178	-3.46	**<0.001**
	Ammonia	1979-2019	1,221	-3.32	**<0.001**
	Nitrate	1979-2019	1,200	-2.01	**0.046**
	Orthophosphate	1995-2019	1,167	-2.64	**0.009**
	Zinc	1995-2004	1,101	-1.73	0.087
5 Old Durham Beck	DO	1973-2019	1,448	-0.30	0.763
	BOD	1973-2015	1,393	-2.41	**0.016**
	Ammonia	1973-2019	1,496	-4.64	**<0.001**
	Nitrate	1973-2019	1,484	6.73	**<0.001**
	Orthophosphate	1977-2019	1,401	4.03	**<0.001**
	Zinc	1974-2006	1,158	-5.43	**<0.001**
	Lead	1979-2002	1,106	-3.71	**<0.001**

Table 2.6 Linear model summaries of changes in key water quality parameters in the Wear catchment (S6-S10). Site numbers increase from downstream to upstream.

Site	Parameter	Periods	*df*	*t*	*P*
6 Deerness	DO	1973-2019	1,494	-0.36	0.721
	BOD	1973-2007	1,408	-2.62	**0.009**
	Ammonia	1973-2019	1,510	-6.30	**<0.001**
	Nitrate	1973-2019	1,460	5.28	**<0.001**
	Zinc	1974-2013	1,260	-4.67	**<0.001**
	Lead	1994-2002	1,103	-3.50	**<0.001**
7 Brancepeth Beck	DO	1991-2004	1,123	1.40	0.164
	BOD	1991-2004	1,146	1.76	0.080
	Ammonia	1991-2004	1,145	-2.78	**0.006**
	Nitrate	1991-2004	1,110	0.23	0.822
	Orthophosphate	1995-2004	1,95	-0.23	0.820
	Zinc	1995-2004	1,100	-1.18	0.243
8 Bedburn Beck	DO	1991-2019	1,248	1.73	0.085
	BOD	1991-2016	1,226	-0.82	0.413
	Ammonia	1991-2019	1,283	-7.23	**<0.001**
	Nitrate	1991-2019	1,207	-4.96	**<0.001**
	Orthophosphate	1992-2019	1,259	-5.66	**<0.001**
	Zinc	1992-2019	1,252	-2.39	**0.018**
9 Stanhope	DO	1973-2012	1,422	-0.28	0.780
	BOD	1973-2007	1,382	-1.34	0.183
	Ammonia	1973-2012	1,446	-2.98	**0.003**
	Nitrate	1973-2012	1,392	-1.99	**0.048**
	Orthophosphate	1974-2012	1,329	-1.39	0.166
	Zinc	1974-2011	1,384	-7.35	**<0.001**
	Lead	1974-2011	1,295	-3.05	**0.003**
10 Rookhope Burn	DO	1976-2019	1,262	2.10	**0.037**
	BOD	1976-2007	1,262	0.74	0.459
	Ammonia	1976-2019	1,307	-2.84	**0.005**
	Nitrate	1976-2019	1,232	-2.24	**0.026**
	Orthophosphate	1978-2019	1,175	-2.90	**0.004**
	Zinc	1978-2019	1,263	-12.34	**<0.001**
	Lead	1978-2019	1,184	-4.36	**<0.001**

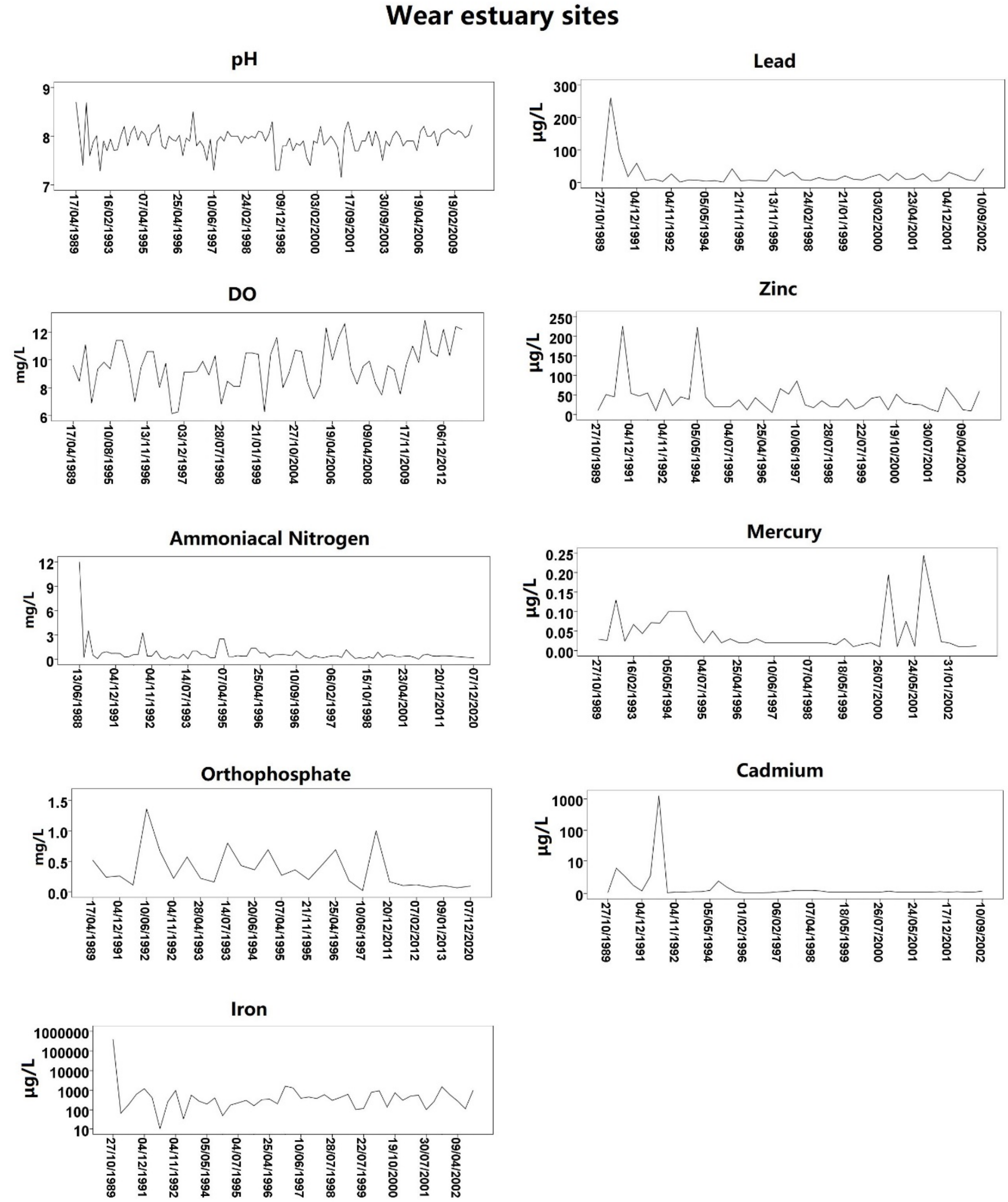


Figure 2.21 Key water quality parameters in the Wear estuary sites from 1988 to 2020 (Wear at South Hylton: 1988 to 2007, 2012, 2013; Wear at Sandy Point: 2008 to 2010; Wear at Copperas Gill: 2011, 2020). Note: Iron and cadmium concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

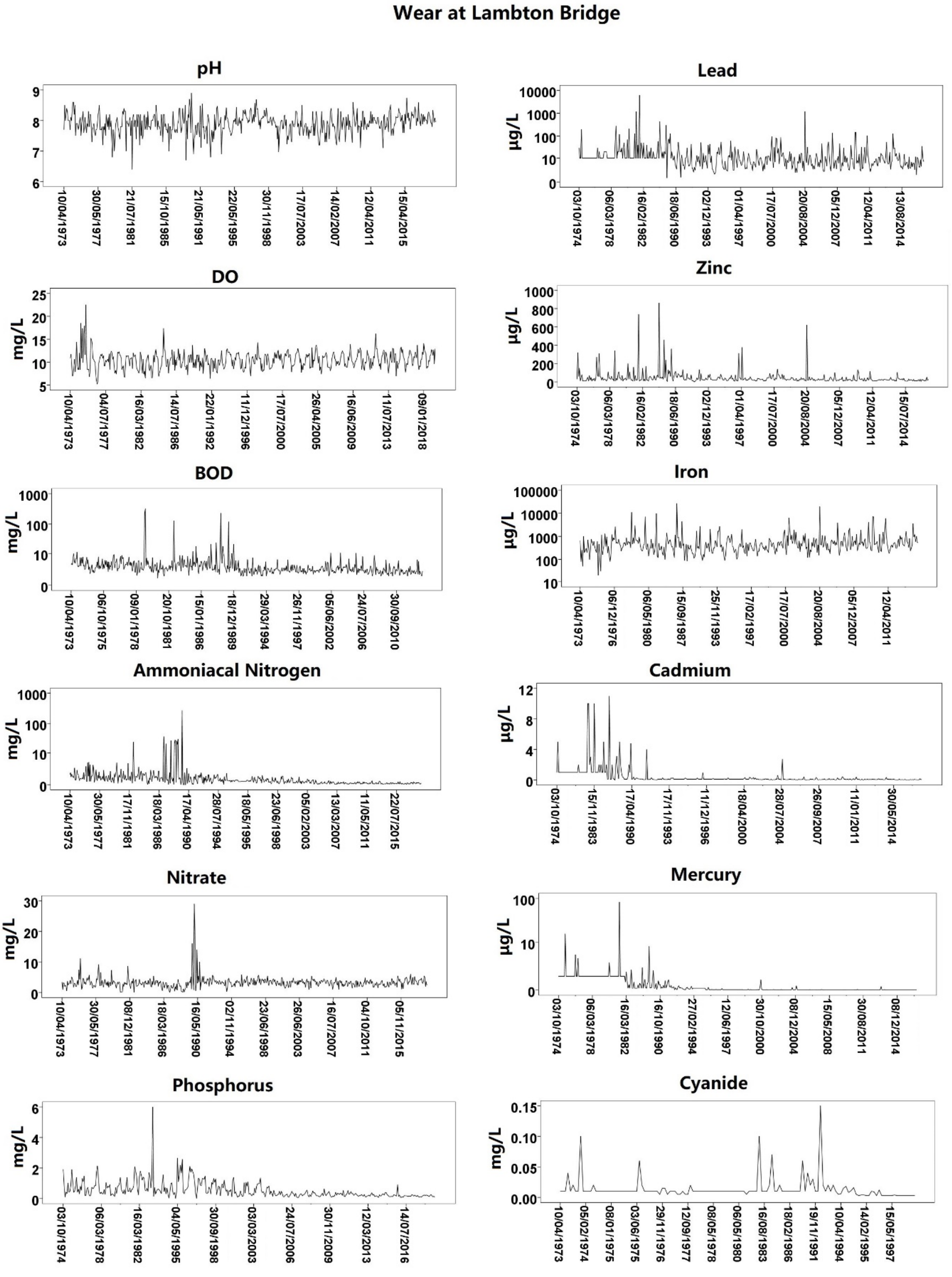


Figure 2.22 Key water quality parameters in the Wear at Lambton Bridge (Chester New Bridge) from 1973 to 2019. Note: BOD, ammoniacal nitrogen, lead, zinc, iron and mercury concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Phosphorus concentrations presented were total P.

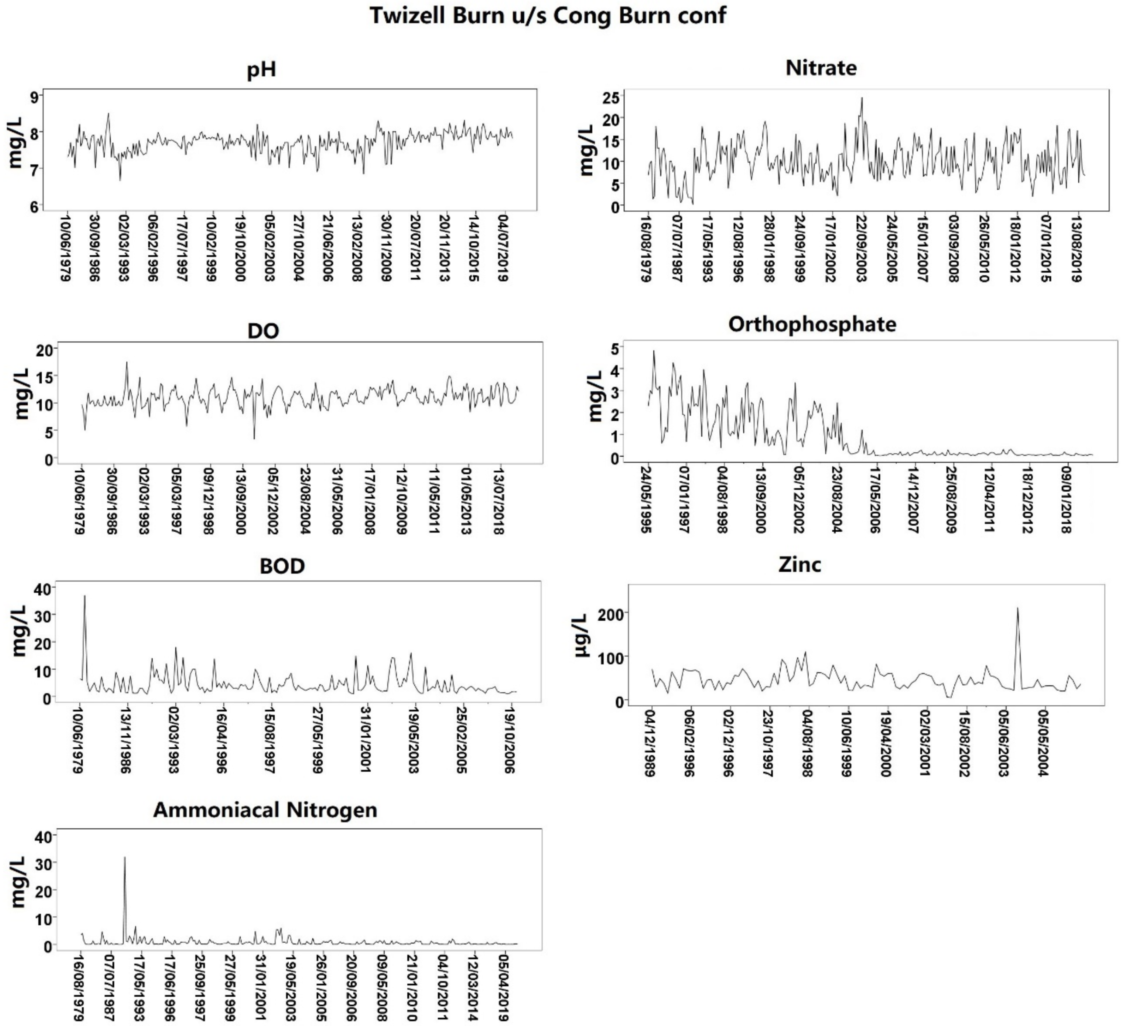


Figure 2.23 Key water quality parameters in the Twizell Burn u/s Cong Burn confluence from 1979 to 2019. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Conf refers to confluence.

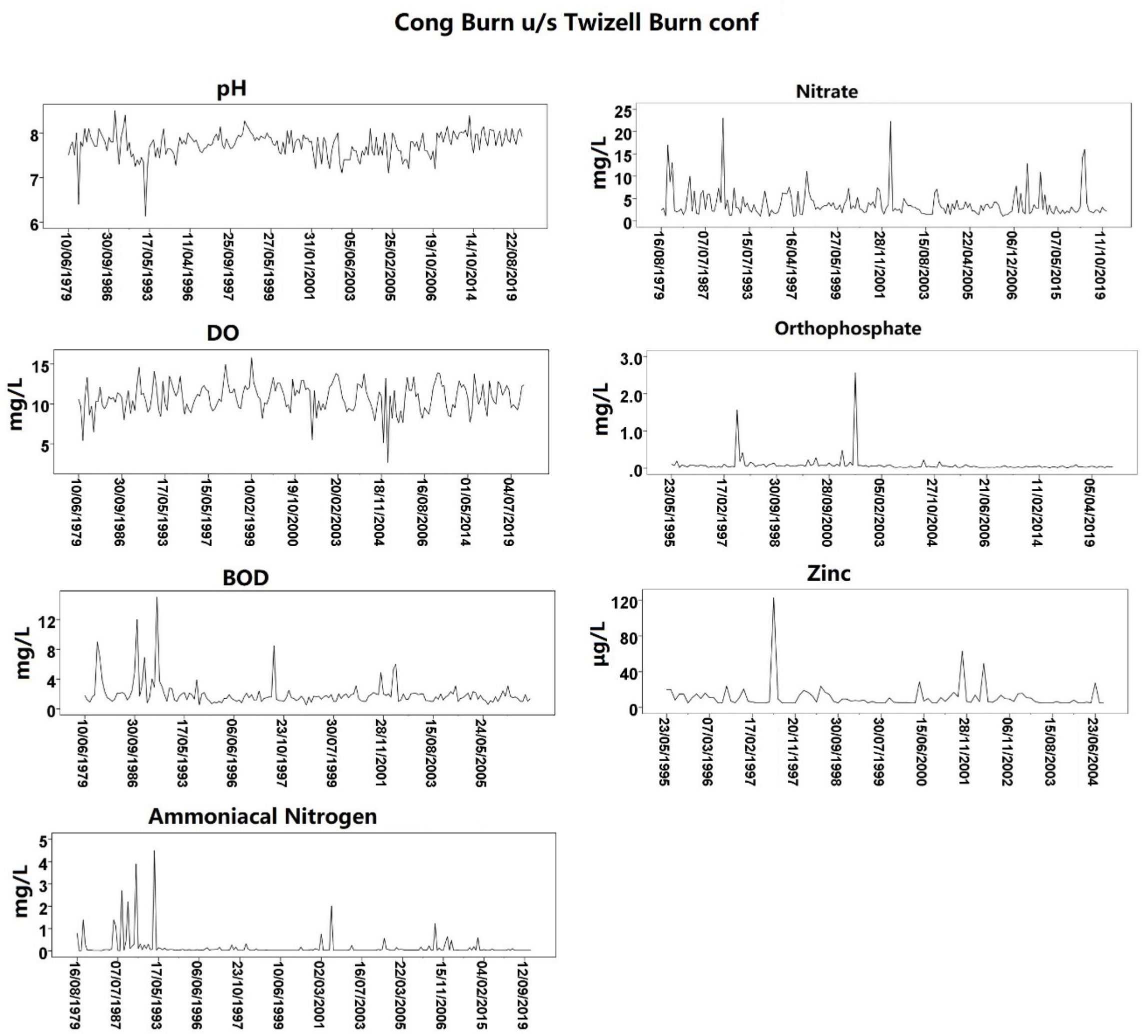


Figure 2.24 Key water quality parameters in the Cong Burn u/s Twizell Burn confluence from 1979 to 2019. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Conf refers to confluence.

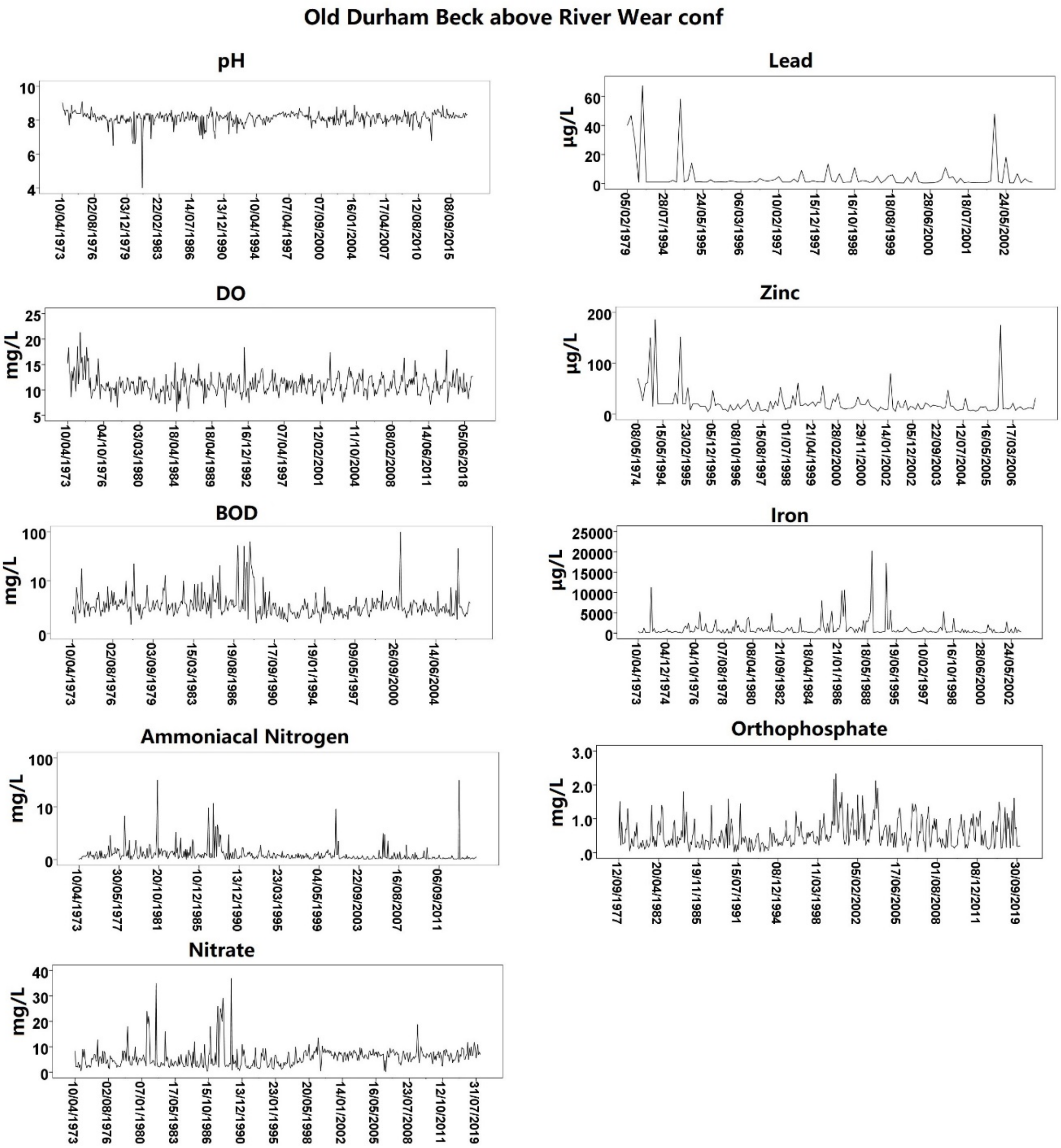


Figure 2.25 Key water quality parameters in the Old Durham Beck above River Wear confluence from 1973 to 2019. Notice: BOD and ammoniacal nitrogen concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered). Conf refers to confluence.

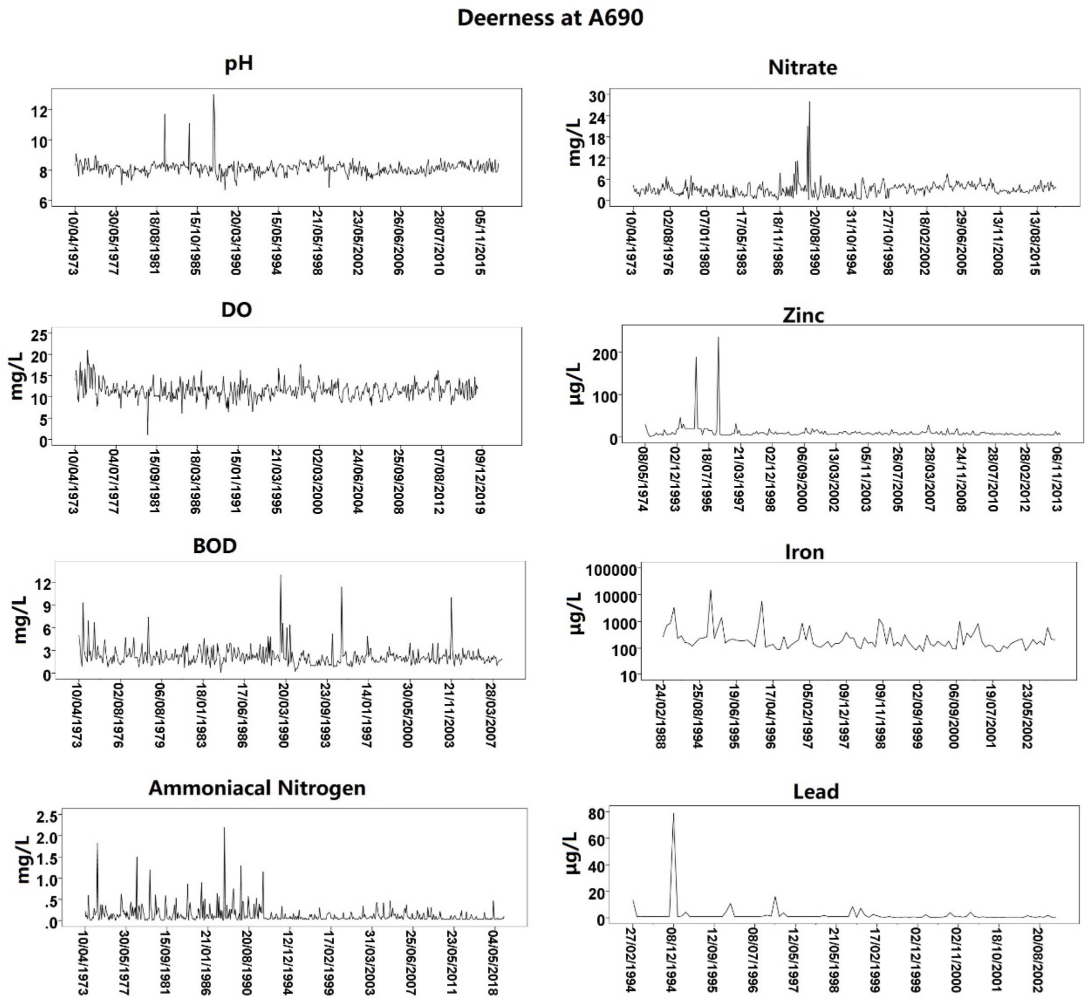


Figure 2.26 Key water quality parameters in the Deerness at A690 from 1973 to 2019. Notice: iron concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

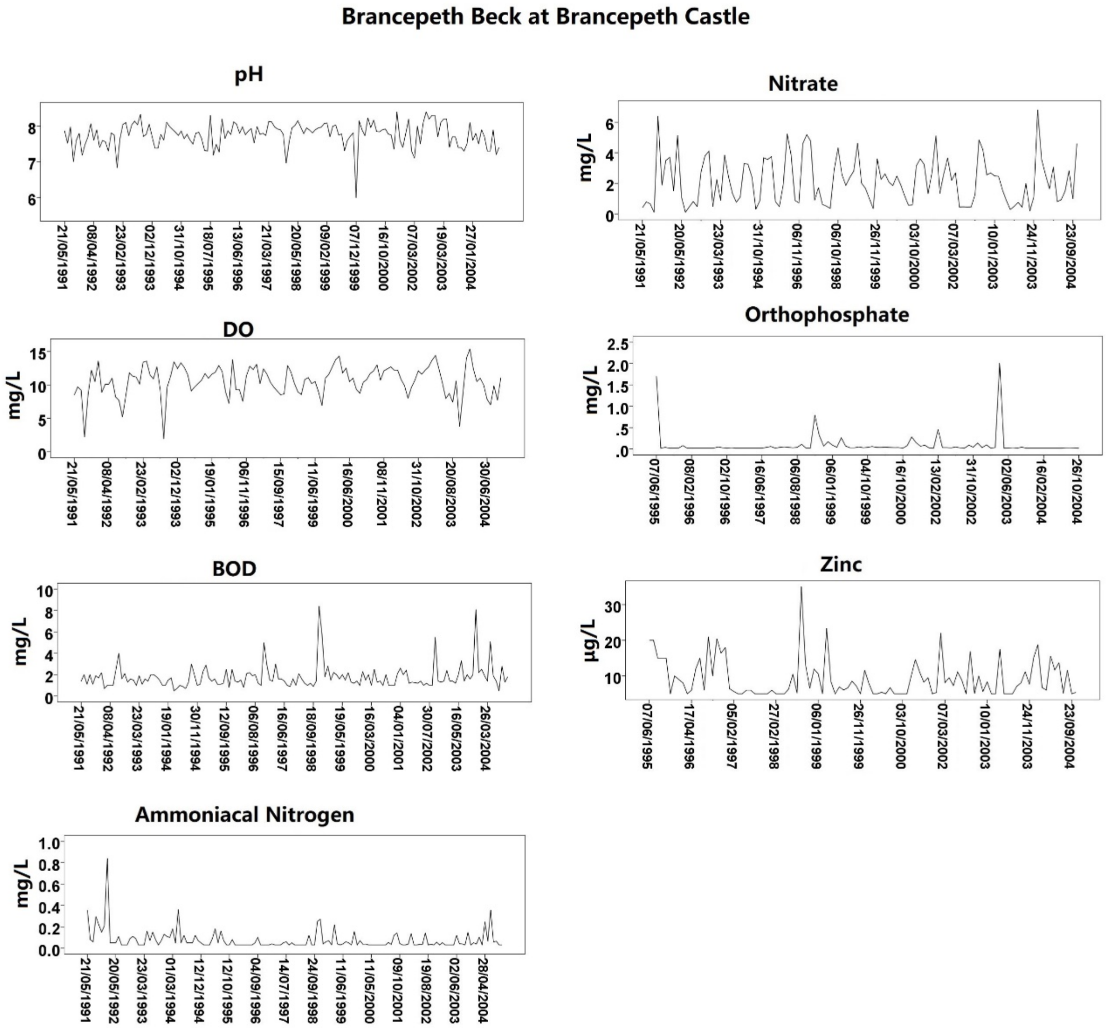


Figure 2.27 Key water quality parameters in the Brancepeth Beck at Brancepeth Castle from 1991 to 2004. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

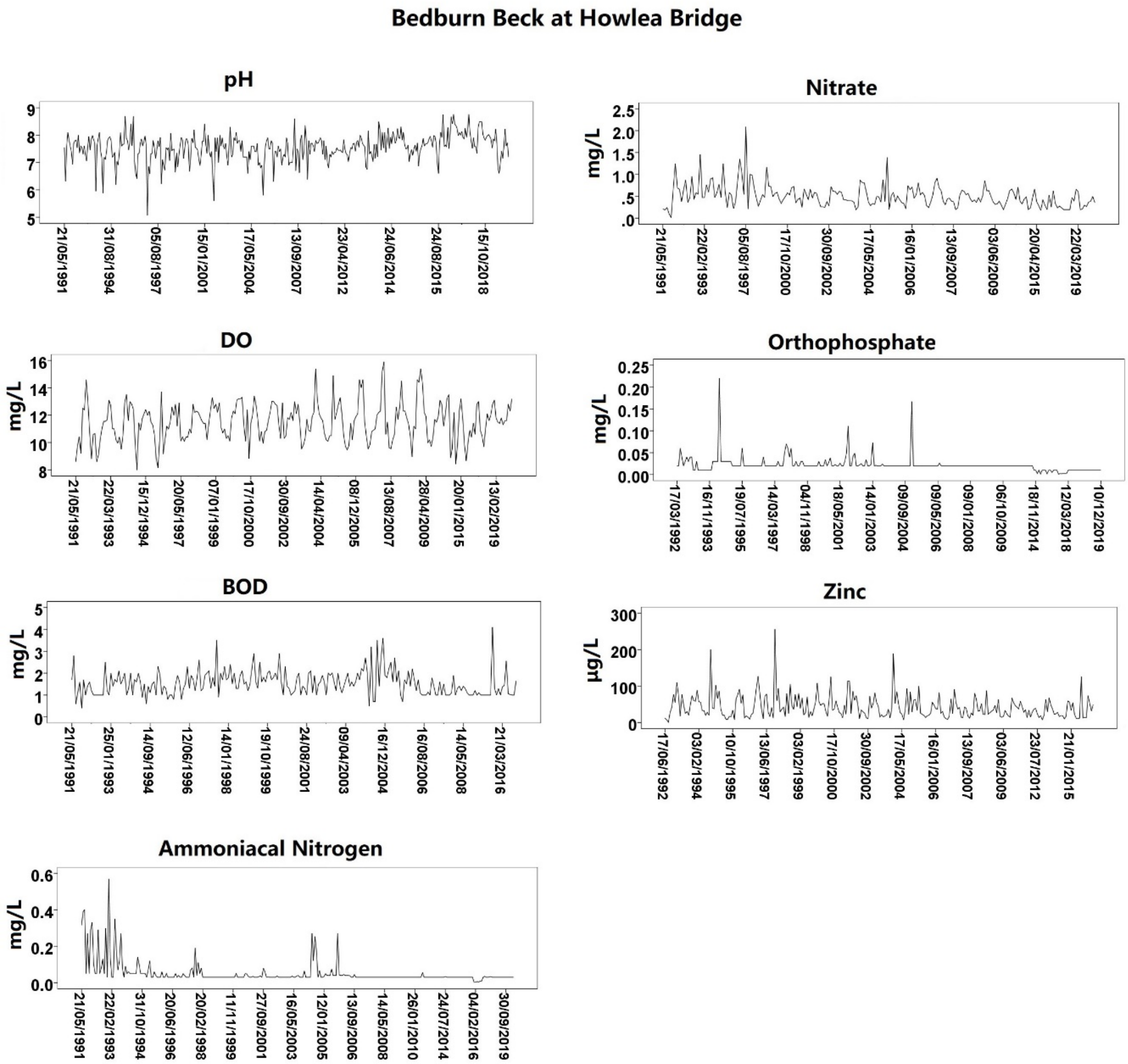


Figure 2.28 Key water quality parameters in the Bedburn Beck at Howlea Bridge from 1991 to 2019. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

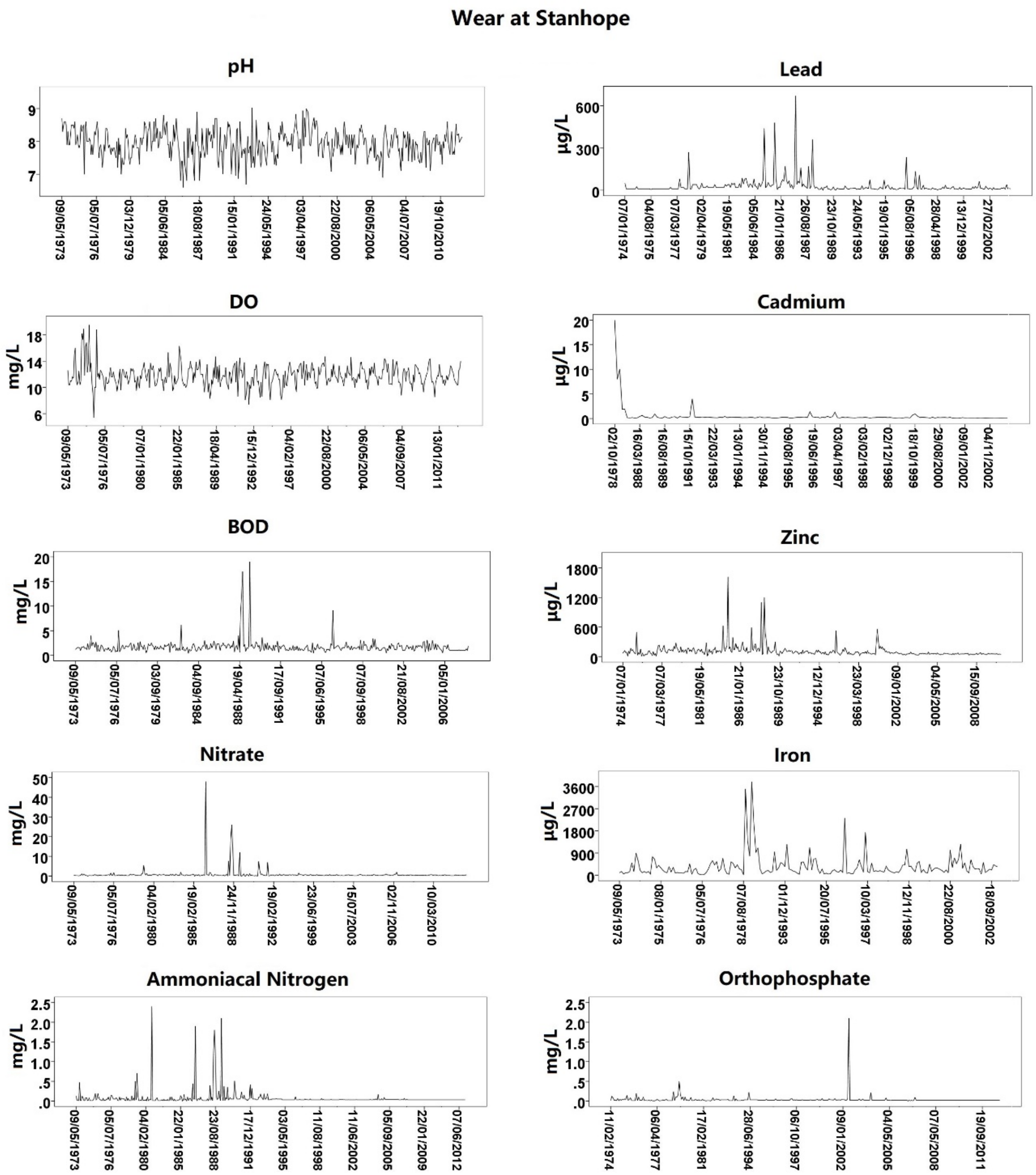


Figure 2.29 Key water quality parameters in the Wear at Stanhope from 1973 to 2019. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

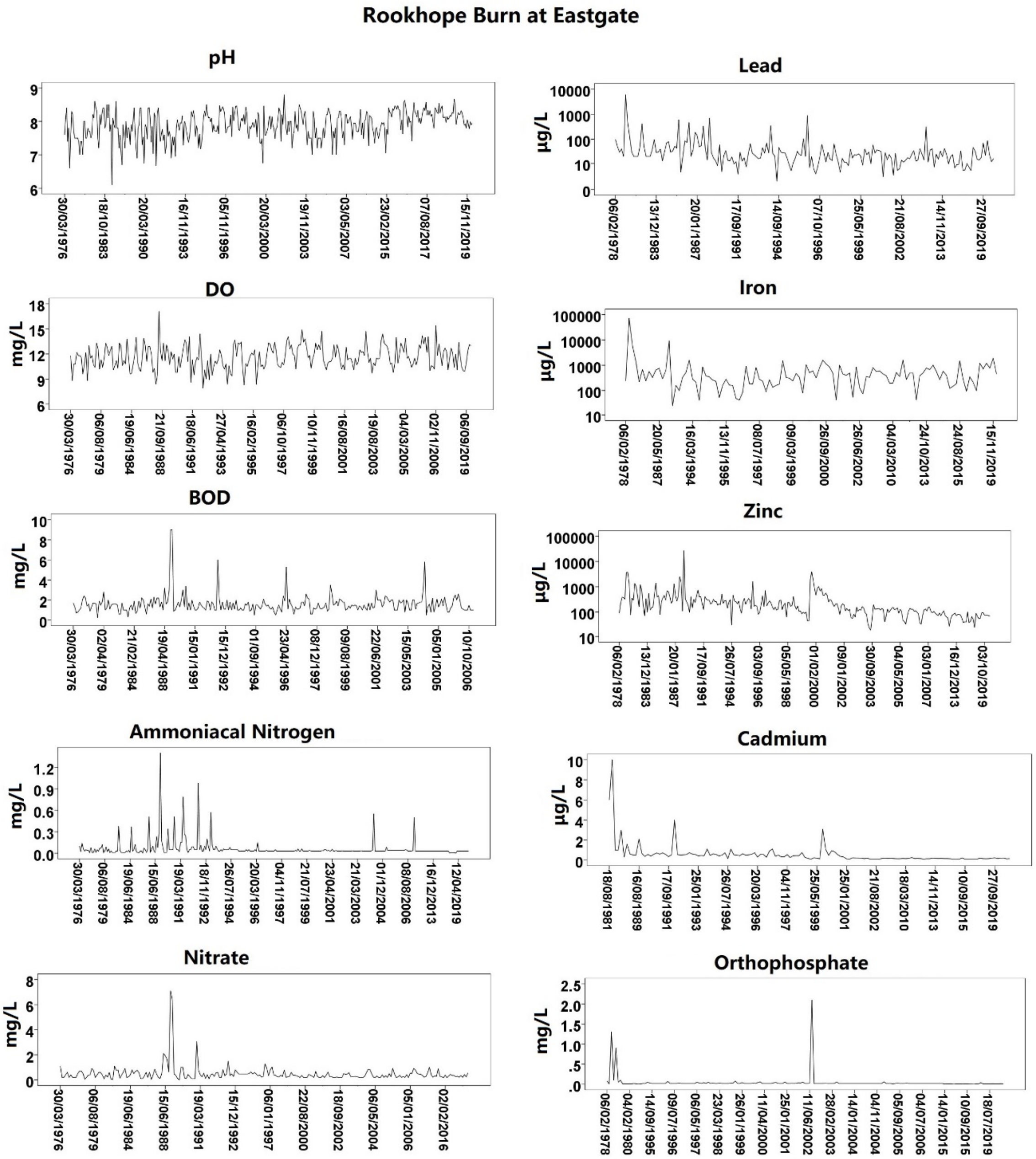


Figure 2.30 Key water quality parameters in the Rookhope Burn at Eastgate from 1976 to 2019. Notice: lead, iron and zinc concentrations are on log scales. Note the different timescales on the panels. All metal element concentrations presented were 'total' values (samples not filtered).

Based on the analysis above it is apparent that chemical water quality has improved across much of the Wear since the 1970s, both for organic pollution indicators and for heavy metals. Similar to the Tyne catchment, the majority of sites for chemical status were 'good' in the Wear in 2015 and 2016 (Table 2.7). However, after new substances were added to the assessment list and stricter standards were developed by the EA, all water bodies in the Wear catchment failed to achieve good chemical status in 2019 (Table 2.7).

No clear changes in the ecological status were found from 2015 to 2019, 58/64 (90.6%) of water bodies still failed to reach good ecological condition in 2019, with the greatest pressures coming from wastewater, urban and minewater pollution sources and from hydromorphological modification (Table 2.8). Evidently, there is still a long way to go in the recovery of the Wear, even if dramatic improvements have been made in recent decades.

Table 2.7 Ecological and chemical classification for surface waters in the Wear catchment in 2015, 2016 and 2019.

Wear catchment		Ecological status or potential					Chemical status	
Year	Number of water bodies	Bad	Poor	Moderate	Good	High	Fail	Good
2015	64	2	10	44	8	0	11	53
2016	64	0	14	44	6	0	8	56
2019	64	1	15	42	6	0	64	0

Table 2.8 Issues in the Wear catchment preventing waters reaching good ecological status under WFD by cycle 2 (2015-2021) and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land management	Domestic General Public	Industry	Local and Central Gov	Mining and quarrying	Recrea tion	Urban and transport	Water Industry	Other	No sector responsible	Sector under investi-gation	Total
Changes to the natural flow and levels of water	-	-	-	-	-	-	-	2	-	-	-	2
Pollution from rural areas	22	-	-	-	-	-	-	-	-	-	-	22
Pollution from abandoned mines	-	-	-	-	34	-	-	-	-	-	-	34
Pollution from waste water	-	-	1	-	-	-	-	55	2	-	-	58
Physical modifications	8	-	-	9	-	2	17	8	1	-	3	48
Pollution from towns, cities and transport	-	1	1	-	-	-	2	30	-	-	-	34
Non-native invasive species	-	-	-	-	-	-	-	-	-	8	-	8

2.3.2.5 Recovery of the Wear fishes

Following the water quality improvement, fish in the Wear catchment started to recover. During beam trawling surveys in the 1980s, 24 fish species were caught (69 1-km trawls) in the Wear estuary, compared to 45 species in the Tyne estuary (880 1-km trawls) and 25 species in the Tees estuary (225 1-km trawls) (Pomfret *et al.*, 1988). Salmon in the Wear have made a clear recovery from the pollution, similar to the Tyne salmon. Salmon rod catches increased on the Wear from the 1970s to the 2010s (Figure 2.15). Very low catches of salmon were recorded between 1965 and 1980, with a number of years recording zero catch. In 1981 there was a peak at 499 before catches decreased and then fluctuated at around 200 until the late 1990s (Environment Agency, 2008a). From the 1990s salmon catch began to increase and reached a maximum of 1,731 in 2013. Sea trout catches on the River Wear followed a similar pattern to salmon. Very low catches of sea trout were recorded in the 1950s. The annual rod catch fluctuated between 88 and 673 from 1960 to 1989. From the early 1990s, total catches of sea trout steadily increased, peaking in 2002 (rather earlier than for salmon), when 2374 were caught. Since 2010, both the salmon and sea trout rod catch have shown a decreasing trend, and the catches in 2017 reduced to nearly half compared with 2010. The rod catch per unit effort (catch per licence day) of salmon in the Wear catchment steadily increased between 1993 and 2008, and then fluctuated between 2009 and 2017 (Figure 2.16). The rod catch per unit effort of sea trout continuously increased between 1993 and 1998, then fluctuated between 1999 and 2017. Some excess salmon from the Kielder programme were stocked in the Wear (Figure 2.17), but numbers were very few, and indicate that the Wear salmon and sea trout recovery was based on natural recolonization, probably facilitated by strays from other river stocks in the region.

In 1992, the National Rivers Authority carried out an electro-fishing survey at 21 sites on the river from Fatfield (near tidal limit) to North Carr Woods (upstream of Bedburn Beck - Wear confluence). These surveys showed the higher reaches of Wear were dominated by salmonid species, especially brown trout, and lower reaches held a mixed stock of coarse fish including a high abundance of dace (*Leuciscus leuciscus*) (National Rivers Authority, 1994d). Subsequent electrofishing, angler and hydroacoustic surveys, overseen by the Environment Agency has shown that the abundance of cyprinid fish, especially dace, in the lower river has decreased in the last decade, possibly as a result of increased numbers of avian fish predators in winter (M. Lucas, pers. comm.).

To monitor and assess the stocks of migratory salmonids in the River Wear, a resistivity fish counter was installed at the salmonid pass at Framwellgate Weir, Durham, located 22 km upstream of the tidal limit. The counter commenced operation in November 1994 (Environment Agency, 2008a). A second counter on a new fish pass at Freemans on the same set of weirs at Framwellgate started operating in February 2015. Total fish counts (salmon and sea trout) fluctuated between 1994 and 2020 (Figure 2.18). The highest total recorded on the Framwellgate Counter was in 1999, with 27,658 fish. From 2017 to 2020, the total count reduced dramatically. This trend, together with rod catch data suggests that adult salmonid numbers may have declined somewhat in the Wear in recent years, although fish counts at Durham are incomplete as fish can ascend the weirs directly in high flows, and prolonged low flow periods in some years can result in reduced river entry and upstream migration.

Since 1991, an annual juvenile salmonid survey has operated across the Wear catchment (Environment Agency, 2008a). Electrofishing surveys were carried out before this by the NRA's predecessors, but these data were not accessible and may no longer exist. Both semi-quantitative surveys (i.e. by which estimates of minimum population abundance are obtained from a fixed area) and timed surveys (by which estimates relative to the fishing method, time or area are obtained) have been undertaken (Environment Agency, 2008a). Further, some conventional quantitative 'depletion' electric fishing surveys have been carried out for a subset of sites and dates. These surveys provide information on the temporal and spatial distribution of the juvenile salmon and trout in the main river and some major tributaries in the Wear catchment. The density both at individual sites and across the catchment can also be classified through the National Fisheries Classification Scheme (Environment Agency, 2008a).

Between 1989 and 1996, the Wear was stocked with juvenile salmon from the Kielder hatchery. A total of 177 thousand juvenile salmon were release into the catchment. However, the numbers are much lower comparing with the fish stocked in the Tyne and Tees. Apart from salmon and trout, the EA also stocked the Wear and some tributaries with chub (*Squalius cephalus*), dace, barbel (*Barbus barbus*) and grayling in the 1990s and 21st century (McParlin, 2011; EA unpublished information). Debate has occurred over the degree to which several of these species are natural to the Wear and should be stocked, even though there is a demand from anglers. In particular, there seems to be good evidence that barbel are non-native to the Wear (Wheeler, 1969; Britton and Pegg, 2011; McParlin, 2011).

2.3.2.6 Changes of fish communities in the Wear tributaries

Three sub-catchments were chosen for assessing long-term fish community change in the Wear catchment (Figure 2.31), concentrating on those studied in Chapter 5. Unfortunately, the data series timescale of these is poor in the Environment Agency's database. In the Cong Burn sub-catchment, three sites were surveyed by the EA by electric fishing (single pass on some occasions, multipass on others – for comparability data are therefore presented as the number captured in first fishing divided by the area [minimum density], supported by capture efficiency estimates where available), the earliest survey dating back to 2003. In the Chester-le-street site, species mainly consisted of brown trout (minimum density range, 0.3 to 38.6 per 100 m^2), stone loach (*Barbatula barbatula*) (minimum density range, 0 to 4.9 per 100 m^2) and European eel (minimum density range, 0 to 5.5 per 100 m^2) (Figure 2.32). Species richness has increased from just two species in 2003 to seven species in 2017 (four in 2019), indicating slow recolonization and diversification of the community by several species. Bullhead (*Cottus perifretum*) and Atlantic salmon were caught in the 2011 and 2017 surveys. In 2017, low abundance of three-spined stickleback (*Gasterosteus aculeatus*) and *Lampetra sp*. were recorded at this site. In 2019, a low abundance of minnow was recorded at this site. Total fish abundance (all species combined) has shown an increasing trend since 2003, although was low in 2019.

In the Browney sub-catchment (excluding the Deerness sub-catchment), 11 sites were surveyed by the EA, using electric fishing (single pass on some occasions, multipass on other occasions) and the earliest surveys date back to 1995 at Malton. As in Cong Burn, there has been a tendency for an increase in species richness in the Browney sites over time with four species at Malton and three at Wall Nook, recorded by 2003, but eight species at the former over 2017-2019 and seven species at the latter over 2012-2016. In 1995, the Malton site was dominated by stone loach (666.7 per 100 m^2), with less abundant brown trout, bullhead and minnow (Figure 2.33). Since 2003, trout (minimum density range, 5.3 to 126.3 per 100 m^2) has become the dominant species, and shifted to minnow in 2019. In the Wall Nook site, between 2003 and 2009, the dominant species was trout (minimum density range, 49.2 to 88.9 per 100 m^2), while bullhead, stone loach, minnow, salmon and three-spined stickleback were present in low densities. From 2010 to 2014, both bullhead (minimum density range, 45.5 to 54.9 per 100 m^2) and minnow (minimum density range, 0.7 to 606.1 per 100 m^2) increased and became co-dominant species. Salmon now seem to be a consistent element of the fauna at Wall Nook (minimum density 2009-2016, 18.7 to 88.3 per 100 m^2). The minnow abundance largely decreased in 2016, no clear changes were found in other species. According to the EA records European river lamprey was occasionally caught in the river, but those records are untrustworthy since the captures are normally larvae or transformers which cannot readily be distinguished between *L. fluviatilis* and brook lamprey (*L. planeri*) (M. Lucas, pers. comm.). By contrast, adult river lamprey have not been recorded or observed upstream of Durham since 1992 (M. Lucas, pers. obs.).

In the Deerness sub-catchment, four sites were surveyed by the EA since 2001 by single-pass electric fishing. The fish community at Langley Moor in 2003 and 2011 mainly consisted of brown trout (minimum density, 7.8 to 50.2 per 100 m^2) and European bullhead (minimum density, 5.6 to 27.8 per 100 m^2), along with lower abundance of stone loach, minnow, and three-spined stickleback (Figure 2.34). In 2017, the abundance of stone loach, minnow, and three-spined stickleback largely increased. At the Ushaw Moor site, the dominant species was brown trout (minimum density, 11.7 to 52.2 per 100 m^2) between 2001 and 2006. Bullhead, minnow and stone loach were present in lower abundance (minimum density, 0 to 11.1 per 100 m^2). European eel, brook lamprey and Atlantic salmon were occasionally caught in the river.

In the Bedburn Beck sub-catchment, six sites were surveyed by the NRA/EA by electric fishing (single pass on some occasions, multipass on others) since 1991. The Newhall Farm site was dominated by salmon (minimum density, 30.4 to 117.3 per 100 m^2) and trout (minimum density, 23.4 to 98 per 100 m^2) between 1991 and 2016 (Figure 2.35). Bullhead were rare at Newhall Farm until 2008 but have been quite abundant since (minimum density, 5.0 to 21.1 per 100 m^2). Lamprey, minnow, stone loach and eel were both present in very low densities and occasionally caught during the surveys. In 2019, minnow abundance slightly increased and it became the dominant species, while salmonid abundance was at its lowest for the whole time series.

Beyond those surveys, some more general comments can be made about the fish communities of the River Wear. River lamprey are moderately abundant downstream of Durham and regularly spawn below Framwellgate weir where samples have been collected in the 2000s for genetic analysis (Bracken *et al.*, 2015) and where hundreds of post-spawned lamprey were collected in the early 1980s for zoological dissections at Durham University (M. Lucas, pers. comm.). Sea lamprey have regularly been observed spawning downstream of Chester-le-Street weir, and upstream to Framwellgate in the 2000s (M. Lucas, pers. comm.). Most of the Wear tributaries with good water quality and some fine sediment as well as gravel, including Bollihope Beck, Waskerley Beck, Shittlehope Beck, Bedburn Beck and the River Browney contain adult breeding brook lamprey (M. Lucas, pers. comm.; Bracken *et al.*, 2015). Dace, chub, roach (*Rutilus rutilus*), gudgeon (*Gobio gobio*) and barbel are most abundant in the main river downstream of Croxdale, although small numbers of barbel, chub and dace are known to occur as far upstream as Bishop Auckland (McParlin, 2011, M. Lucas pers. comm). Grayling are most abundant from Durham to Wolsingham (M. Lucas, pers. comm.).

Flounder are found throughout the tidal river.

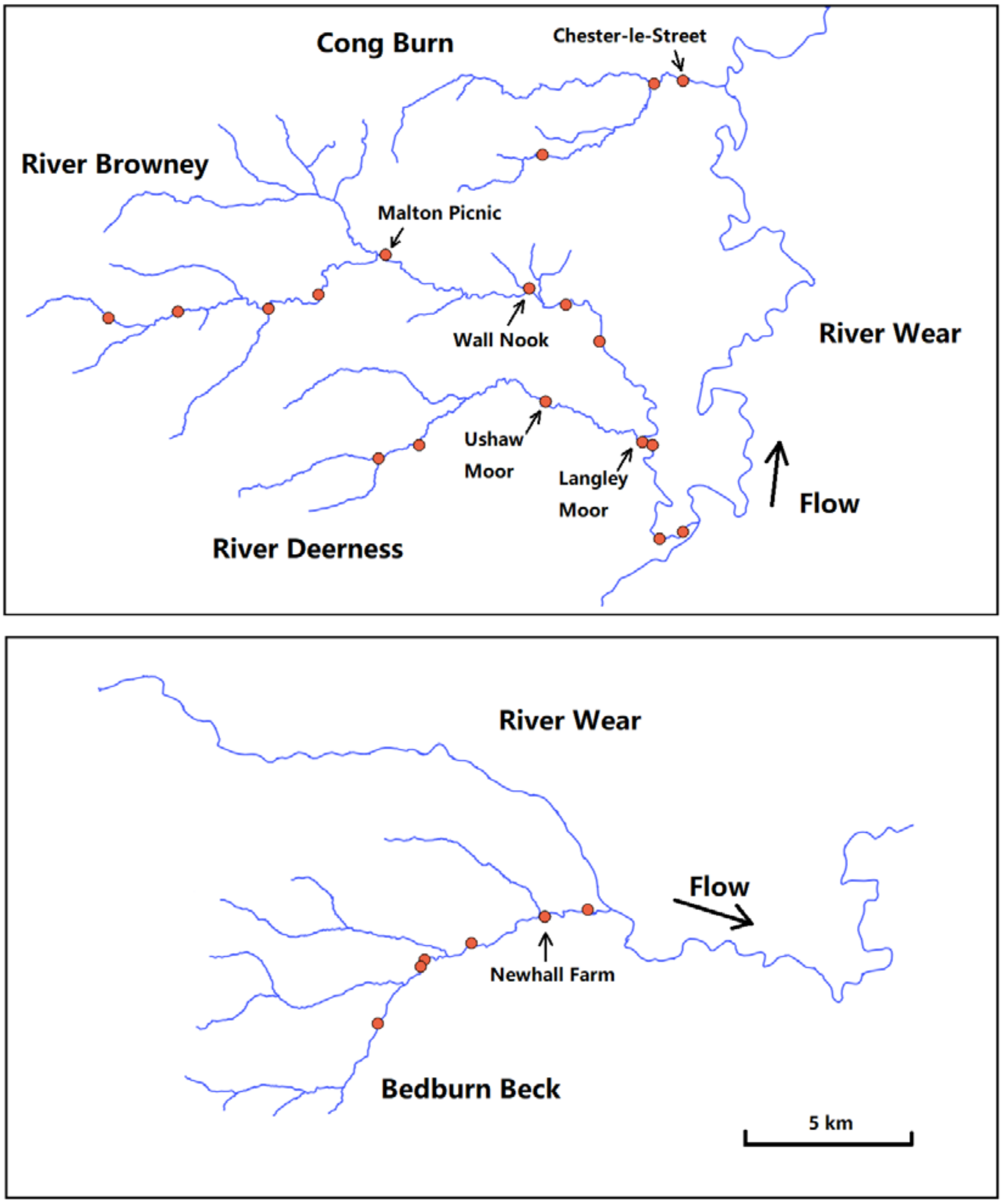


Figure 2.31 Location of EA fish sampling sites in selected Wear tributaries for which data is presented below. Sites without names represent EA sites with much shorter time records, which were not included in this study.

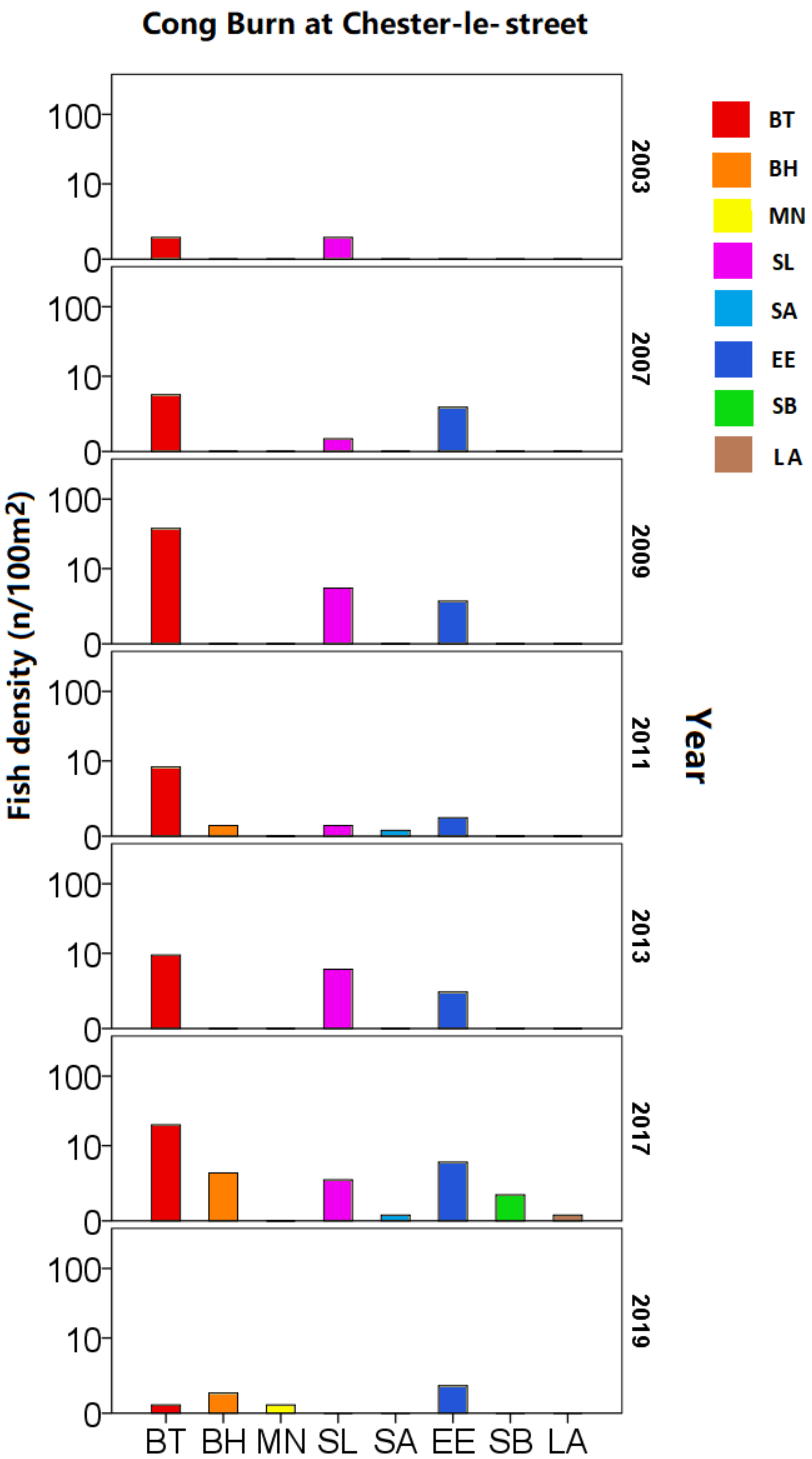


Figure 2.32 Long-term variation of estimated minimum fish density (2003-2017: single pass electric fishing; 2019: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 63.6%, bullhead 84.6%, minnow 100%) at Cong Burn. BT: brown trout, BH: bullhead, MN: minnow SL: stone loach, SA: Atlantic salmon, EE: eel, SB: three-spined stickleback, LA: *Lampetra sp*.. Note the log scale.

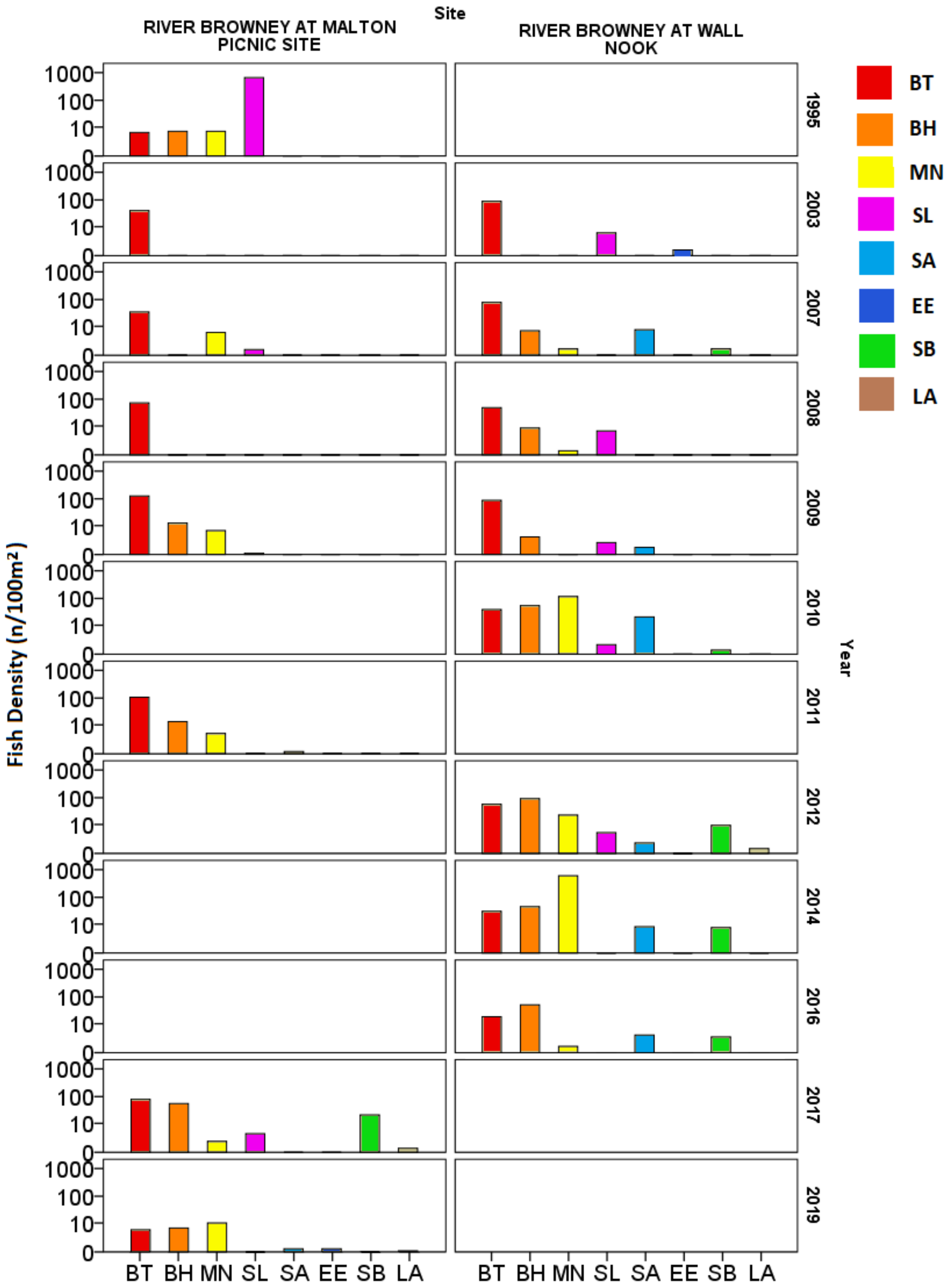


Figure 2.33 Long-term variation of estimated minimum fish density (1995-2017: single pass electric fishing; 2019: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 84.5%, bullhead 87.0%, minnow 76.6%) at River Browney. BT: brown trout, SA: Atlantic salmon, BH: bullhead, SB: three-spined stickleback, LA: *Lampetra sp.*, MN: minnow, SL: stone loach, EE: eel. Blank panel means the site was not surveyed. Note the log scale.

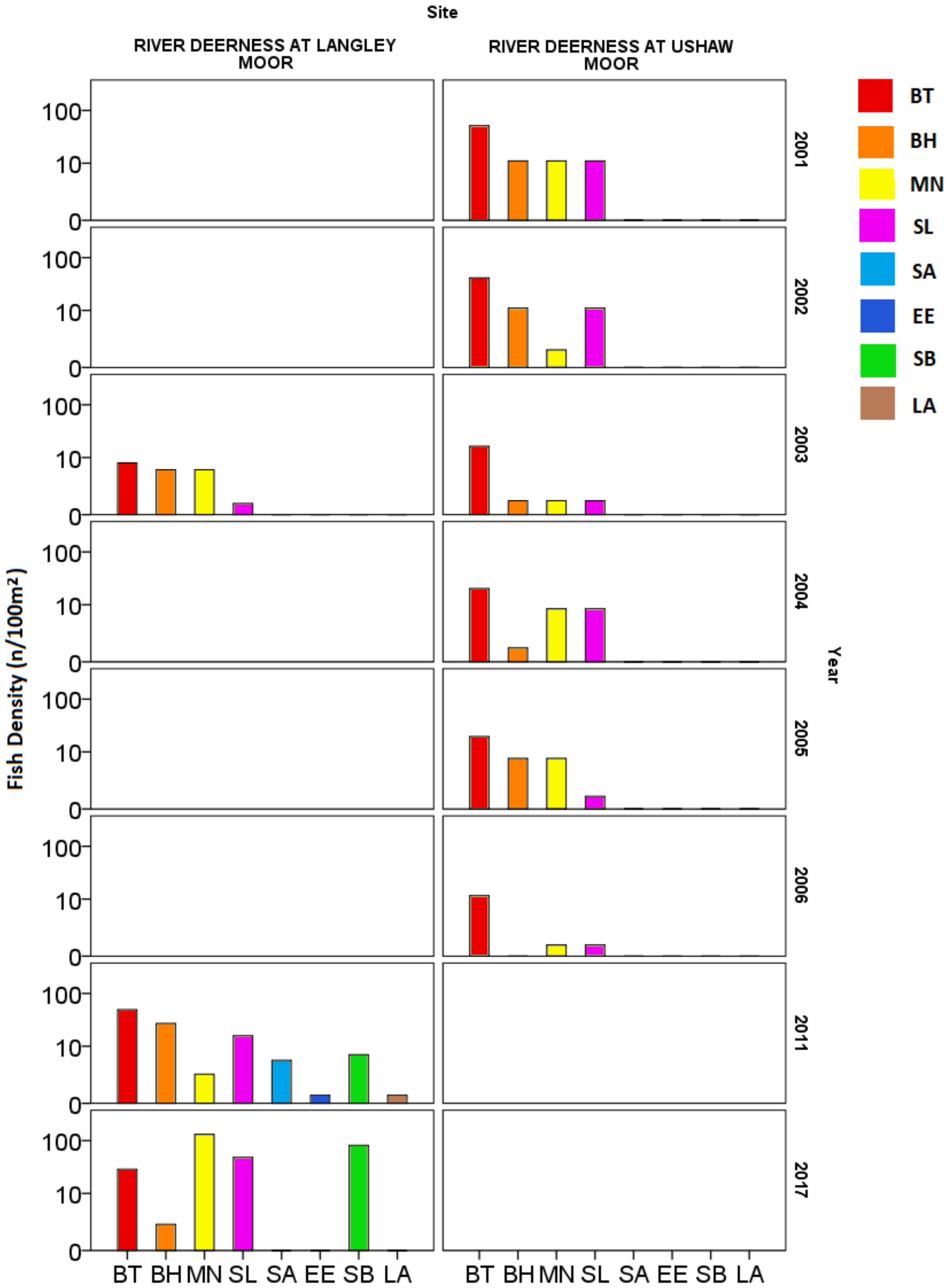


Figure 2.34 Long-term variation of estimated minimum fish density (single pass electric fishing) at River Deerness. BT: brown trout, SA: Atlantic salmon, BH: bullhead, SB: three-spined stickleback, LA: *Lampetra sp.*, MN: minnow, SL: stone loach, EE: eel. Blank panel means the site was not surveyed. Note the log scale.

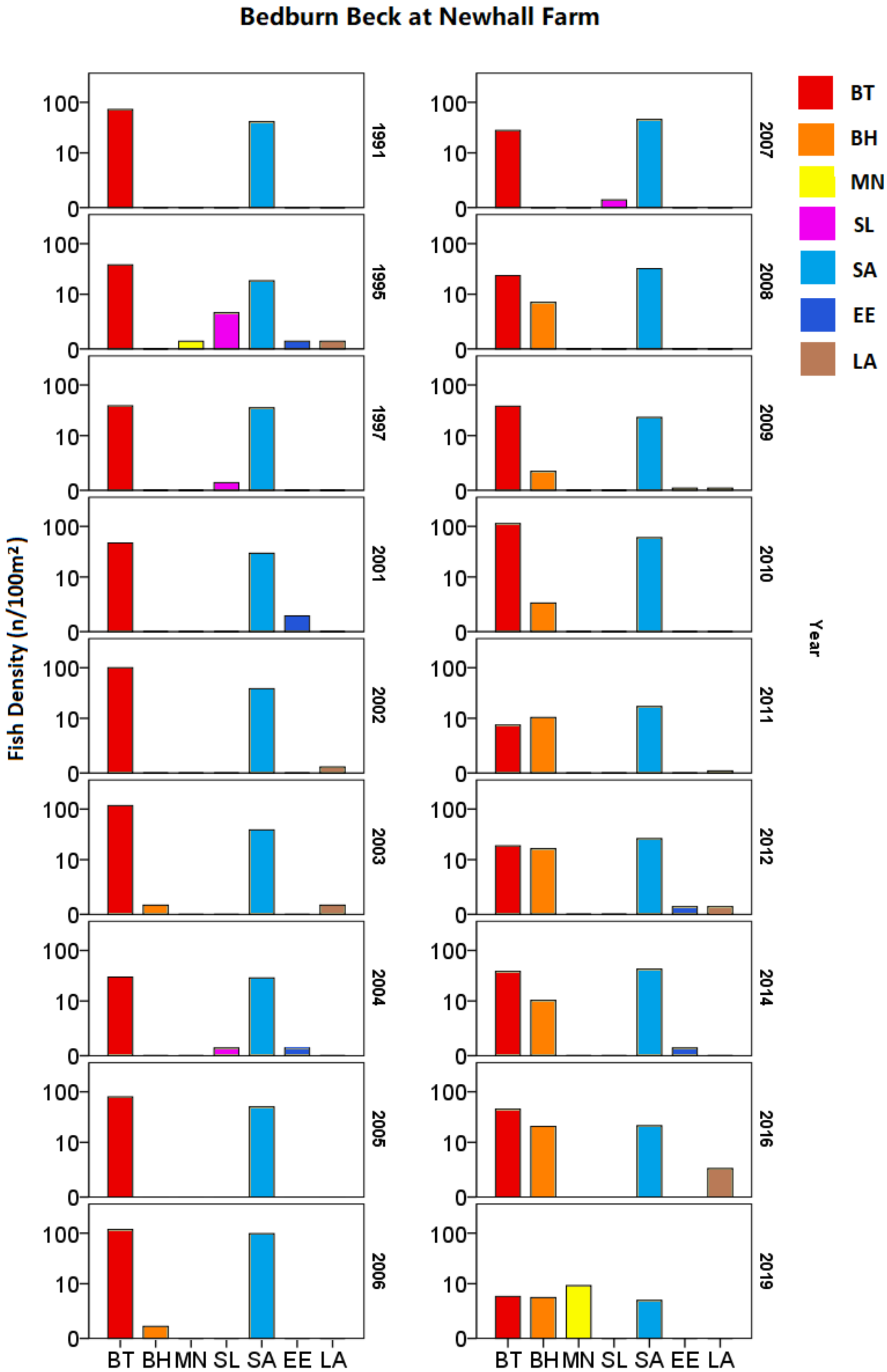


Figure 2.35 Long-term variation of estimated minimum fish density (1991-1997, 2007-2016: single pass electric fishing; 2001-2006, 2019: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 93.7%, salmon 84.2%, bullhead 60.6%, minnow 77.0%) at Bedburn Beck Newhall Farm site. BT: brown trout, SA: Atlantic salmon, BH: bullhead, LA: *Lampetra sp.*, MN: minnow, SL: stone loach, EE: eel. Notice: the 1991 survey only counted trout and salmon. Note the log scale.

2.3.3 River Tees

2.3.3.1 History of Tees fishes

The River Tees, used to be a major salmon river like the Tyne and Wear. Some of the earliest fish records there go back to the early 15th Century, where numerous fisheries and fishgarths (fish weirs) were for lease in the Tees (Anonymous, 1928). Fisheries were an important activity in the Tees region, and the majority of fish taken from the river were salmon. Apart from salmon, smelt (*Osmerus eperlanus*) used to be abundant in the Tees, and they were commercially exploited in the Tees estuary during the 16th Century (Maitland, 2003). In 1530, regulations were made to deal with disagreements between fishermen using draw nets (seines) and haling-nets (framed nets on poles). An order was made in that year that fishing with 'kydyll' nets for smelt and herring (possibly referring to shad *Alosa sp.*) was prohibited, that the fishing season for smelt would be 25 April to 1 August, and smelt could only be taken from upstream of the Saltholme (Brewster, 1796; Anonymous, 1928).

In the 18th Century, the salmon fishery was the principal fishery in the Tees catchment. Large amounts of salmon were caught in Stockton, and after the town's needs were catered for, the remainder were sold to York, Leeds and other cities (Anonymous, 1928). Salmon was allowed to be taken from the 22nd of November to the 12th of August according to the act of parliament issued by George I (Brewster, 1796). A large drag net was used by fishermen for catching salmon between the estuary and Dinsdale-lock. The weir at Dinsdale was about 2.4 m high with two locks (not for navigation), one on each bank, and fish could only ascend it during the flood period (Figure 2.36; Commissioners for the British fisheries, 1861). In the shallow downstream reach, fishermen would also use spears to catch salmon from boats. During the spawning season, both salmon and sea trout were protected so they could reach the Barnard Castle reach and further upstream to spawn (Brewster, 1796). Harbour seals (*Phoca vitulina*) at one time were numerous in the Tees, and the population was estimated at about 1,000 in the Tees Estuary in the early 19th Century (Anonymous, 1928; Wilson, 2001). Because they preyed on the salmon, it was the custom for local fishermen to devote a day or two occasionally to hunting the seals to protect the fishery (Anonymous, 1928). Apart from salmon, the Tees also produced great numbers of flounder, eel and smelt in the estuary reach (Parson, 1827). However, due to the large supply of sea fish, these were rarely to be found in the market (Parson, 1827). From Middleton-One-Row to further upstream, the river also produced a good abundance of brown trout in the shallow reach (Brewster, 1796).

Figure 2.36 A view of the fish locks and weir on the river Tees at Dinsdale in the early 19th Century, drawn by J.M. Sparks.

2.3.3.2 The decline of Tees fish and other fauna

In the 19th Century, the River Tees was still an important salmon river with around 10,000 fish being netted from the river in 1867 (Netboy, 1968). Subsequently the salmon run was ruined by the combination of industrial pollution and in-stream barriers. In Teesside, following the opening of railways in 1825, the coal trade in Stockton increased greatly. The dramatic industrial development led to a population explosion in Middlesbrough between 1830 and 1840 (Warren, 2018). Teesside become one of the major industrial centres in Britain, with the pollution that came with it.

Like the South Tyne and the upper Wear, the upper Tees catchment was badly polluted by heavy metals due to the increased hushing and lead ore washing (Commissioners for the British fisheries, 1861). The upper Tees was suggested to be one of the most affected rivers by historical metal mining (Hudson-Edwards *et al.*, 2008). In the middle reaches, the major tributary, the Skerne, was heavily modified as a result of industrialization, urbanization and land drainage. The middle reach of the Skerne was straightened and channelized between 1850 and 1945, and the river corridor was markedly narrowed, then channel widening and deepening was undertaken in the 1950s and the 1970s to protect the housing and infrastructure from flooding (Vivash *et al.*, 1998). In addition, the river was polluted by sewage from Darlington (the main town along the freshwater course of the

Tees) and chemical works discharges in the mid-19th Century, and fish mortalities due to pollution were observed (Commissioners for the British fisheries, 1861). In-stream barriers also caused significant impacts to the fish community, Dinsdale weir was considered to be the major man-made obstruction to fish in the Tees, which blocked the upstream salmonid migration most of the time. Additionally, there were numerous mill weirs constructed further upstream above Dinsdale weir (e.g. Whorlton lock) and the majority of them were installed with fish traps (Commissioners for the British fisheries, 1861).

Pollution problems were partly resolved by the end of the 19th Century, enabling net catches of 5,000 to 9,000 salmon between 1905 and 1916 (Environment Agency, 2009b). Then, a major decline began in the early 20th Century and by the middle of the century the stock of salmon had decreased almost to the point of extinction (Watson and Davis, 1995). From the source to Middleton-in-Teesdale, no significant organic pollution occurred, and there was little variation in chemical composition (Anonymous, 1938a), though heavy metal contamination must have been evident. Between Middleton-in-Teesdale and Croft (where the Skerne tributary joins) the effluents from several small sewage works were discharged into the river (Anonymous, 1938a), but the quantity of sewage effluent was relatively small and it was considered that it did not cause harmful effects to the river (Anonymous, 1938b, 1938a). From the river's middle reaches downstream, the River Tees suffered major pollution due to urbanization and industrialization. At Croft, the entry of the River Skerne caused a marked change in the chemical and biological characteristics of the Tees (Anonymous, 1938b, 1938a). The Skerne was heavily polluted with sewage effluent from the town of Darlington, its water was frequently devoid of dissolved oxygen and led to a complete loss of fishes (Anonymous, 1938a). Although pollution occurred, the non-tidal reaches of the lowerTees still held substantial numbers of brown trout and other species such as grayling, dace, chub, gudgeon and roach (Anonymous, 1938a).

The lower catchment and estuary reach was particularly affected by pollution as a result of industrialization and urbanization (Watson and Davis, 1995). During and after the Great War, there had been a considerable increase in the number of coke-ovens (Sheail, 2000). Pollution discharges mainly consisted of untreated sewage discharges from Middlesbrough and Stockton, coke-oven effluents and pickle liquors (a strong acidic solution) from steel works (Southgate *et al.*, 1932). Approximately 4400 lb of tar acids and 1800 lb of cyanide were discharged into the estuary on a daily basis (Alexander *et al.*, 1936). Substantial numbers of dying smolts in the estuary were reported by the Tees Fishery Board in 1920 (Brady *et al.*, 1982; Rushall, 1996). Considerable numbers of adult salmon and sea trout were recorded dead while attempting to pass through the estuary during both upstream and downstream migrations (Southgate *et al.*, 1932; Alexander *et al.*, 1936; Anonymous, 1938a), and led to a great decline in the value of the salmon and sea trout fishery (Anonymous, 1938a). In 1930, about 600 salmon and sea trout smolts were killed or poisoned in the estuary during downstream migration (Southgate *et al.*, 1932). It was suggested that cyanide was the main factor responsible for the death of smolt in the estuary (Southgate *et al.*, 1932). In the spring of 1931, 66 dying salmon and 131 dying sea trout smolts were picked up from the estuary, and their gill colours were compared with the colours generated from those exposed to varying concentrations of cyanide (Bassindale *et al.*, 1933). The gill colour of dying Tees smolts in the estuary was brighter than for normal fish, indicating a characteristic symptom of cyanide poisoning (Bassindale *et al.*, 1933; Alexander *et al.*, 1936). Apart from salmon and sea trout, dead whiting (*Merlangius merlangus*) and other fish were found on almost every tide in the Tees estuary in late 1926 (Sheail, 2000). By 1935, salmon was virtually extinct in the Tees, and the invertebrate fauna distribution in the Tees estuary was greatly influenced by pollution (Shillabeer and Tapp, 1989; Tapp *et al.*, 1993). It was suggested that the middle reach of the estuary was totally deoxygenated and high concentrations of phenol and cyanides from coke-oven effluents to the estuary were directly toxic to fish (Shillabeer and Tapp, 1989).

The severe estuarine pollution along with the construction of numerous weirs and other in-stream barriers on the river, eventually resulted in the complete loss of both salmon and sea trout from the catchment between the 1930s and the early 1980s (Moore and Potter, 2014). Like salmon and sea trout, the smelt population also decreased dramatically during the same period and became extinct in the Tees (Maitland, 2003). In 1926, Imperial Chemical Industries (ICI) was formed and produced large-scale investment in the chemical complex at Billingham (Nelson, 2003), and expanded to the Wilton and North Tees sites in the late 1940s (Ord, 1988). Their production included fertilizers, heavy organic chemicals and chlorine (Nelson, 2003). In 1970, the total BOD load to the estuary was estimated at 500 tonnes per day (Ord, 1988). In the late 1980s, the chemical industry overtook the steel industry on Teesside and became the dominant industry in the area (Shillabeer and Tapp, 1989). These chemical works have contributed large volumes of waste water to the estuary. Apart from industrial pollution, the river was also affected by disposal of domestic sewage (Nelson, 2003). Untreated sewage of 400,000 people was discharged to the Tees estuary directly until the Portrack sewage works were completed in 1985 (Brady *et al.*, 1982).

Apart from fixed pollution sources, multiple pollution incidents were recorded in the Tees catchment since the 1980s, indicative of periodic damage. In particular, the frequent occurrence of these in the 1980s to early 2000s in rivers such as the Skerne inhibited ecological recovery. In the upper Tees, a pollution incident occurred approximately 1 km upstream of High Force on the 24 October 1983, when nearly 3000 gallons of flux oil leaked into the river from Hargreaves Quarriers (Owen *et al.*, 1993). It was estimated 3000 dead fish were removed from the polluted reach and the total number of fish was many times higher (Owen *et al.*, 1993). In September 1997, an unidentified pollution incident killed approximately 90 dace in the River Skerne at Barmpton (Jenkins, 1998). On 24 February 2000, an unidentified substance entered the Skerne at Aycliffe reach, this incident led to a massive fish kill from the entry point all the way to the Skerne-Tees confluence (Jenkins, 2000). More than 1180 dace, 800 chub, 35 trout and small numbers of roach, pike, perch (*Perca fluviatilis*), gudgeon and grayling were killed during the incident (Jenkins, 2000).

Apart from salmon and sea trout, the previously abundant mammal, the harbour seal, completely vanished from the Tees estuary during the Industrial Revolution in the mid-19th Century (Wilson, 2001). The seal population started to decline following land reclamation in the early 19th Century and the increase of iron trade in Cleveland and shipping industries in Middlesbrough (Wilson, 2001). The Tees estuary shore has been heavily modified for petrochemical and other heavy industrial use, and the upper reaches of the estuary have been canalized (Shillabeer and Tapp, 1989). In addition, the development of Teesport and increased dredging activities in the estuary in the 1960s led to further losses of large intertidal areas (Shillabeer and Tapp, 1989). The area of intertidal mudflats has been reduced by more than 90% (Wilson, 2001; Smurthwaite, 2006). By the 1960s and early 1970s, seals were rarely recorded in the Tees catchment due to habitat loss (Environment Agency, 1999c). In addition, sediments in the Tees estuary were heavily polluted by metal and organochlorine contaminants, including highly elevated concentrations of zinc, copper, lead, cadmium, chromium and mercury compared to less industrialized rivers such as the River Bure and Yare in Great Yarmouth (Smurthwaite, 2006).

In the upper Tees catchment, the headwaters of the Tees and two tributaries (River Balder and River Lune) have been dammed to create reservoirs. Cow Green Reservoir is the only reservoir on the main River Tees. Construction of Cow Green dam began in 1967 and the dam was closed on 1 June 1970 (Crisp, 1977). The River Lune has two reservoirs, the upstream one is Selset reservoir which flows into Grassholme reservoir. The River Balder has three reservoirs, with the largest being Balderhead reservoir, it flows into Blackton reservoir then into Hury reservoir before reaching the River Tees (Nelson, 2003). These reservoirs are operated by NWL to provide drinking water and industrial water for Darlington and Teesside, and regulate and augment river flow to support extraction and release compensation flow during dry seasons (Nelson, 2003). As explained in Section 2.3.1.3 the steel and chemicals industry at Teesside used very large amounts of water and its development and predicted increase in water need was a primary reason for development of Kielder Reservoir in the Tyne valley and the Tyne-Wear-Tees interbasin water pipeline, as well as Cow Green Reservoir. Fish population surveys provided evidence that below Cow Green Reservoir, trout and bullhead densities increased after the dam construction (Crisp *et al.*, 1983). Benthic invertebrate surveys immediately below the Cow Green dam in 2004 indicated that nineteen of the thirty-one common taxa in the regulated Tees reach declined in numbers, including *Hydra sp.*, *Ancylus fluviatilis*, Naididae, Heptageniidae, *Leuctra fusca*, *Leuctra inermis* and *Brachycentrus subnubilus* (Armitage, 2006). However, some taxa including *Lymnaea peregra, Ephemerella ignita, Hydroptila sp.* increased in numbers (Armitage, 2006). In addition, the reservoirs along the River Lune and Balder not only eliminated upstream spawning habitat, but also prevented the downstream transport of coarse substrate, which reduced the salmonid spawning habitat downstream (Environment Agency, 2010).

2.3.3.3 The Tees barrage

In 1990, the Teesside Development Corporation proposed a barrage across the Tees which was enabled through an act of Parliament. The purpose of the barrage was to control the flow of the river, preventing flooding and the effects of tidal transport of pollutants upstream. It was closely linked to planned business and leisure regeneration along the banks of the Tees in Stockton and Thornaby, popularized in Prime Minister Margaret Thatcher’s “Walk in the Wilderness” on Teesside in 1987. Construction work started in November 1991, the barrage was completed in 1994 and opened in April 1995. The Tees barrage represents the most recent of a large series of human impacts on the estuary (Tapp *et al.*, 1993). After construction was completed, the length of estuary was reduced to 18km and the upstream section of the barrage was turned into a freshwater reach (Environment Agency, 1999c). The barrage determines the upstream limit of the estuary and saline intrusion (Environment Agency, 1999c). The barrage has four hydraulically operated gates that can be lowered to allow flow over, and was designed to incorporate a whitewater kayak facility on the left bank (Figure 2.37). The barrage also has a navigation lock on the right bank.

To allow adult salmon and sea trout to pass upstream of the barrage, a Denil fish pass was installed next to the north bank pavilion and an elver pass was also installed to facilitate the upstream passage of juvenile eels (Watson and Davis, 1995). The flow through the pass was relatively small but the attractiveness of its entrance was intended to be supported by the whitewater course outflow. The River Tees Barrage and Crossing Act 1990 did not stipulate a required fish passage efficacy but the operators must demonstrate its adequacy to the satisfaction of the EA (not yet attained in 2020). A degree of conflict has existed during the period of barrage operation in the allocation of river flow for fish passage (M. Lucas, pers. comm.). One or more barrage gates (typically the left-most gate) have been used to support passage of jumping adult salmonids by lowering the gate, especially around high tide. But historically, water flow for operating the whitewater course has been prioritized. In 2010 four Archimedes Screws were added to the kayak course for pumping water upstream and generating power, during which time a further upstream fishway was added between the Archimedes Screws.

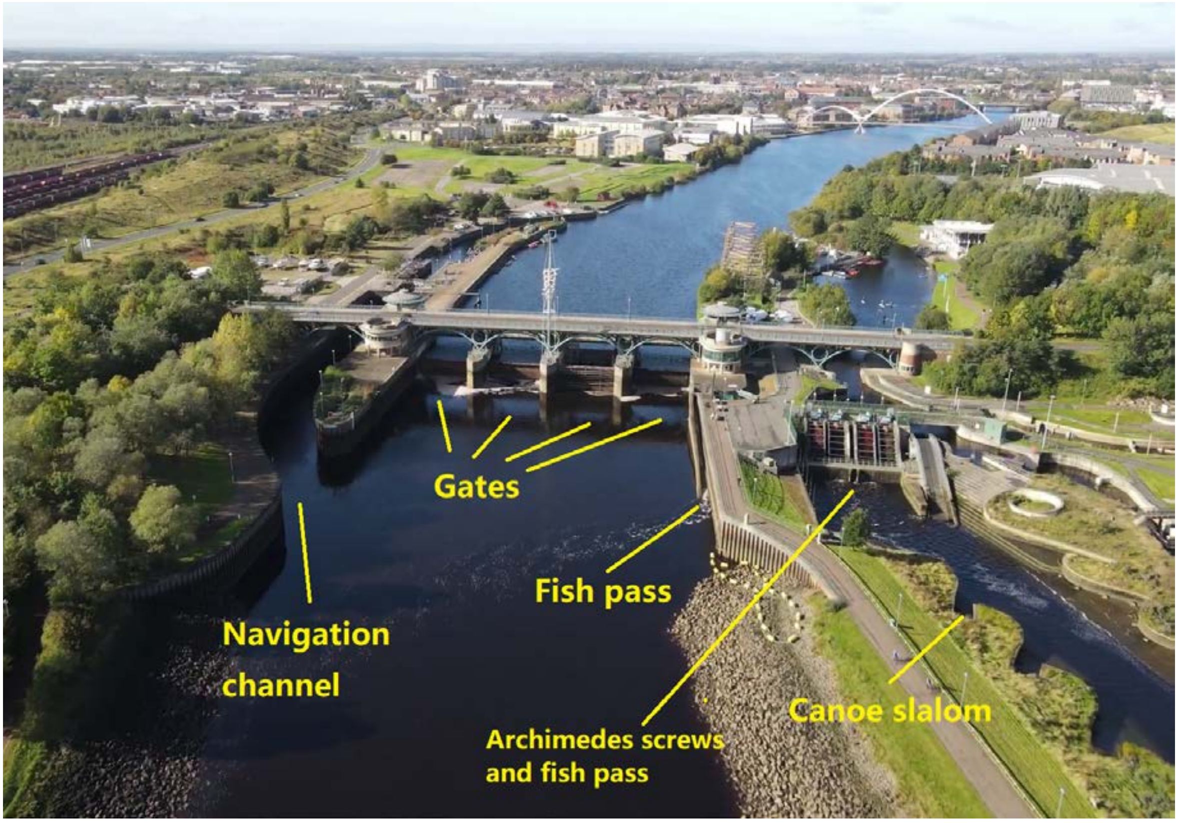


Figure 2.37 The Tees Barrage viewed from downstream (Photo credit: PremiumP UAV).

Between 1995 and 2011, a conventional upstream adult salmonid trap and a resistivity counter was operated at the top of the Denil fish pass at the barrage (Moore and Potter, 2014). In addition, a CCTV camera was installed to try and monitor fish ascending the barrage gates (Rushall, 1996). In 1995, 256 adult salmonids were recorded passing the barrage through the fish pass (Lucas, 1995). In addition, more than 4600 eel, 500 dace, 150 river lamprey and a few flounder were also recorded in the fish pass between 25 May 1995 and 31 October 1995 (Lucas, 1995; Rushall, 1996). Some salmonids were observed to pass the barrage by jumping over the left-hand gate when the tide was high enough, during elevated freshwater flows, and some salmon and sea trout have been found using the navigation lock to pass the barrage (Lucas, 1995; Rushall, 1996).

In 2002, a fish tagging and tracking experiment was carried out by Cefas. A total of 156 fish (103 salmon and 53 sea trout) were caught from the lock, then acoustic tagged and released back downstream. Only three salmon and two sea trout managed to pass the barrage, one passed through the canoe slalom and the other four passed by the barrage gates (Environment Agency, 2009b). In 2008, 2009 and 2013, another fish tracking experiment was conducted by Cefas. A total of 237 fish (199 salmon and 38 sea trout) were captured in T-nets at the estuary mouth, tagged with acoustic transmitters and released at the capture site. A total of 80 fish (33.7%) approached the barrage after release and 11 fish managed to pass the barrage, 10 by the barrage gates and one by the fish pass (Moore and Potter, 2014). Over 30% of tagged fish are known to have died and at least 14.7% were predated by seals (Moore and Potter, 2014). In recent years resident seals, especially grey seal (*Halichoerus grypus*) have become an increasing cause of adult salmonid mortality at the barrage, highly conspicuous to onlookers as those seals eat salmon close to the barrage. Long passage delays and repeated passage attempts due to the barrage's obstructing effect put the salmonids at high predation risk.

The number of sea trout caught in the trap at the fish pass increased rapidly from 1995 to 2001 then declined until 2011 (Environment Agency, 2009b; Moore and Potter, 2014). The number of salmon caught each year by anglers in the Tees was relatively stable between 1995 and 2003, peaked in 2004 (439 fish), and was variable until 2011. After the previous tracking studies and video showed that the fish trap was causing fish to move back downstream, the trap was removed and replaced with an updated resistivity fish counter (Moore and Potter, 2014), operated by the EA since July 2011.

Apart from salmon and sea trout, a monitoring program of the eel elver run was undertaken on 11 occasions in 1996. A total of 1455 elvers were caught in the elver pass between May and August, suggesting the elver pass facilitated upstream migration of elvers (Environment Agency, 1999c), though this number is a fraction of one percent of the expected run size. Also, sub-adult 'yellow' eels have occasionally been recorded during the beam trawl surveys throughout the estuary and the cooling water intake at Hartlepool power station in the lower Tees (Environment Agency, 1999c).

2.3.3.4 Recovery of the Tees habitat

Since the 1970s, the metal industries in the Tees Estuary have shown a decline (Smurthwaite, 2006), including the loss of most of the steel industry, with the massive SSI Redcar steelworks finally closing in 2015 (Evenhuis, 2018). The chemicals industry on

Teesside, concentrated on Billingham, Wilton and Seal Sands, has also slowly declined in recent decades (Evenhuis, 2018). In 1972, Stockton Borough Council drafted a proposal to control domestic sewage pollution (Nelson, 2003). Large interceptor sewers were built to channel discharges from Stockton, Norton and Billingham in the north and from Acklam, Linthorpe and Thornaby in the south, to a newly constructed treatment works at Portrack (Nelson, 2003). From 1973, numerous changes were conducted to improve the water quality of discharges to the Tees estuary (Tapp *et al.*, 1993). Primary sewage treatment was introduced to the major sewers on both sides of the river, many industrial plants have closed and treatment was introduced at some sites (Tapp *et al.*, 1993). By 1996, it was expected that pollution levels in the estuary has reduced 90% compare with 1970 (Watson and Davis, 1995).

Due to the development of North Sea oil and the pipeline link between Teesside and the Ekofisk field, along with the increase of new chemical products, discharges to the estuary increased slightly in the mid-1980s (Warwick *et al.*, 2002). Following strengthened controls on discharges and the introduction of improved treatment measures, both BOD and ammonia concentrations in the Tees estuary decreased (Warwick *et al.*, 2002). The BOD monitoring data showed a major reduction during the 1970s, then the lower BOD levels were continued during 1980s and 1990s (Tapp *et al.*, 1993; Warwick *et al.*, 2002). Following water quality improvement, the benthic fauna inhabiting the area between the estuary mouth and Middlesbrough Dock became more diverse between 1979 and 1990 (Tapp *et al.*, 1993). Significant decreases of Zn, Cu and Cr were recorded in the estuary (Davies *et al.*, 1991), and decreased Pb and Zn levels in the surface sediments were recorded since the 1970s (Jones and Turki, 1997). By 1996, there were 41 sewage treatment works in the Tees catchment, including 15 in the Tees upstream of the Skerne confluence, eight in the Skerne sub-catchment and seven in the Leven sub-catchment (Nelson, 2003).

Harbour seal returned to the Tees estuary in the 1980s, the only known case of recolonization of harbour seal in an estuary from which it had been extirpated (Wilson, 2001). Four counts of 9-17 seals were recorded in June-July 1984 and six counts of 12-22 seals recorded between June and October 1986 (Wilson, 2001). Between 1989 and 1997 harbour seal numbers in the Tees increased from about 24 to 50 individuals (Wilson, 2001). However, this species has suffered from organochlorines (especially polychlorinated biphenyl compounds - PCBs) and heavy metal contamination from industrial chemicals which resulted in a high death rate of pups in the 1980s and 1990s (Wilson, 2001). In 2018, the highest maximum count of harbour seals was of 112 individuals in July, and for grey seals was 45 individuals in August (Bond, 2018).

In the Tees catchment, eight sites were chosen to study the long-term chemical water quality trends (Figure 2.38), employing sites with the longest time series available. Five sites were located in the main river Tees and three sites were located in tributaries. The Tees at The Gares site was located in the river mouth, and reflects the change of chemical trends in the estuary area. The Tees at Newport Bridge and Tees Barrage sites located within the tidal reach, reflects the change of chemical trends in the lower main river. Data from these two adjacent sites were stitched, in order to get a more continuous data run. Tees at Low Hail Bridge site represents the middle reach of the main river. The Tees at the Egglestone Abbey represents the upper main river. For tributary sites, one was located in Greatham Creek, a small tributary running from the north through mudflats and joining the Tees estuary. This river was potentially polluted due to intensive farming and wastewater treatment outflows. This sub-catchment incorporated a study site in which empirical work was undertaken in Chapter 4. One site was located in the River Skerne, which has been heavily modified as a result of urbanisation. One site was located in Clow Beck, which was characteristic of a rural middle reaches lowland stream, but where in the last 50 years neighbouring land has become increasingly intensively farmed.

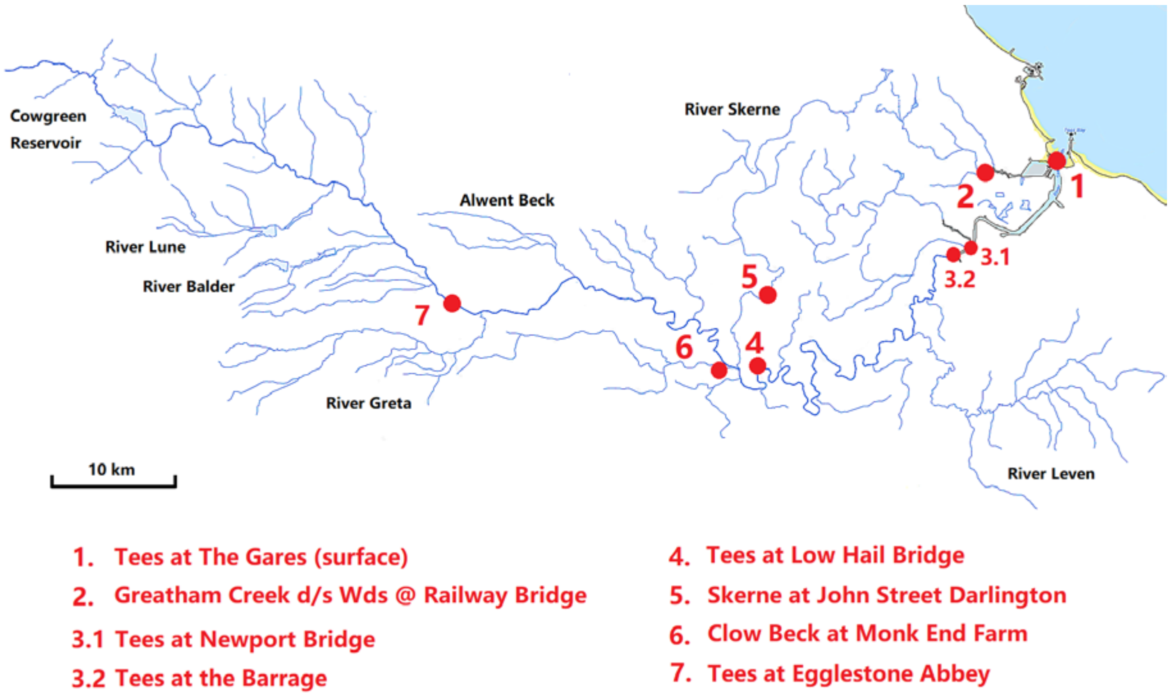


Figure 2.38 Tees catchment and sites with long-term chemical trends for which data is presented in the thesis.

At the estuary site (1989-2019, period varying across determinands), DO, ammonia, nitrate, orthophosphate, zinc and lead all declined significantly over the periods for which data were available (Table 2.9). The decrease in DO is surprising; both pH and DO fluctuated widely before 1999, then became more stable between 1999 and 2019 (Figure 2.39). Ammoniacal nitrogen concentration showed a clear decreasing trend since 1995, with a mean concentration of 1.7 mg/L before 1995 to 0.3 mg/L after 1995 (Figure 2.39). Lead and cadmium concentrations varied considerably between 1989 and 1992, then largely reduced and became relatively stable after that. Mercury concentration declined and became relatively stable (Figure 2.39). Zinc concentration varied considerably between 1989 and 2007, but has fluctuated less and remained around a baseline since 2007 (Figure 2.39).

At the Greatham Creek site (1984-2002), BOD, ammonia, nitrate, zinc and lead all decreased significantly (Table 2.9). pH value varied considerably from two to eight between 1984 and 1997, then became relatively stable after that (Figure 2.40). BOD, ammoniacal nitrogen and nitrate concentrations were varied widely between 1984 and 1997, then dramatically reduced after that (Figure 2.40). Concentrations of all metal showed a strong decline between 1984 and 1997 (Figure 2.40).

At the Newport Bridge/Tees Barrage (downstream of barrage) sites (1980-2020), DO increased significantly and ammonia decreased significantly over the period of records (Table 2.9). pH values stayed stable during most periods, but two peak values were observed between late 1989 and early 1993 (Figure 2.41). DO concentration varied considerably between 1983 and 1996, with frequent hypoxia events, then increased and fluctuated less since 1997 (Figure 2.41). Ammoniacal nitrogen concentration varied considerably between 1980 and 1994, then declined greatly and became relatively stable after that (Figure 2.41). A few peaks of cyanide concentrations were observed between 1989 and 1990, and the measurement increased to 0.5 mg/L between 2015 and 2016.

At the Low Hail Bridge site (1973-2019), BOD, ammonia, nitrate, orthophosphate, zinc and lead all decreased significantly over the periods of data availability (Table 2.9). High concentrations of BOD and ammoniacal nitrogen were observed from the late 1970s to early 1990s, then dramatically reduced after that (Figure 2.42). Nitrate concentration were relatively low between 1973 and 1987, then a few peak values were observed between late 1987 and 1990, then reduced back to previous levels after that. Lead and cadmium concentrations declined greatly since the late 1970s (Figure 2.42). Mercury concentration declined since the late 1980s. Zinc showed a small progressive trend of decline (Figure 2.42). No clear trends were observed in other chemical components.

At the River Skerne site (1973-2019), DO increased and BOD, ammonia, nitrate, orthophosphate, zinc and lead decreased significantly (Table 2.10). pH values were stable most of the time, but extreme values were observed between late 1987 and early 1988 (Figure 2.43). The lowest value of 1.6 appeared in December 1987 and highest value of 11.6 appeared in March 1988. A steady decline in BOD concentration occurred between 1973 and 2007, with an associated increase in oxygen levels over that period also (Figure 2.43). A dramatic decline was observed in ammoniacal nitrogen concentration, which started since 1990. Orthophosphate concentration was variable before 2006, then decreased and became relatively stable after that (Figure 2.43). Iron and cadmium concentration were variable before 1990, then decreased and became relatively stable after that. Mercury concentrations also decreased (Figure 2.43).

At the Clow Beck site (1976-2019), BOD, ammonia, nitrate, orthophosphate, zinc and lead all decreased significantly over the respective data periods (Table 2.10). pH values varied markedly before 1990, then became relatively stable after that (Figure 2.44). BOD, ammoniacal nitrogen and nitrate concentrations were very variable before 1991, then declined markedly and became relatively stable after that (particularly for BOD and NH_3) (Figure 2.44). No clear patterns were apparent for DO and metal elements.

In the Tees at Egglestone Abbey Bridge (1984-2019), BOD, ammonia, nitrate, orthophosphate, zinc and lead all decreased singnificantly over the data timescales (Table 2.10). Peak values of ammoniacal nitrogen and nitrate were observed between 1989 and 1992, then decreased to stable low levels between 1992 and 2019 (Figure 2.45). Other water quality parameters showed no clear trends.

Table 2.9 Linear model summaries of changes in key water quality parameters in the Tees catchment (S1-S4). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea.

Site	Parameter	Periods	*df*	*t*	*P*
1 Estuary	DO	1993-2019	1,203	-2.42	**0.016**
	Ammonia	1990-2012	1,269	-9.46	**<0.001**
	Nitrate	1992-2019	1,216	-2.38	**0.018**
	Orthophosphate	1992-2019	1,330	-3.40	**<0.001**
	Zinc	1989-2013	1,181	-5.46	**<0.001**
	Lead	1989-2011	1,173	-2.77	**0.006**
2 Greatham Creek	DO	1997-2002	1,59	1.31	0.194
	BOD	1984-2002	1,78	-3.01	**0.004**
	Ammonia	1984-2002	1,79	-4.37	**<0.001**
	Nitrate	1984-2002	1,65	-4.80	**<0.001**
	Orthophosphate	1997-2002	1,89	-1.27	0.206
	Zinc	1984-2002	1,68	-4.03	**<0.001**
	Lead	1984-2002	1,68	-9.32	**<0.001**
3 Newport Bridge	DO	1983-2020	1,252	6.93	**<0.001**
	Ammonia	1983-2012	1,361	-14.46	**<0.001**
	Nitrate	1983-2020	1,322	-1.52	0.130
4 Low Hail	DO	1973-2013	1,410	-1.36	0.175
	BOD	1973-2008	1,391	-5.52	**<0.001**
	Ammonia	1973-2013	1,446	-7.56	**<0.001**
	Nitrate	1973-2012	1,421	-4.15	**<0.001**
	Orthophosphate	1978-2012	1,267	-3.47	**<0.001**
	Zinc	1974-2013	1,252	-7.41	**<0.001**
	Lead	1978-2003	1,101	-3.31	**0.001**

Table 2.10 Linear model summaries of changes in key water quality parameters in the Tees catchment (S5-S7). Site numbers increase from downstream to upstream.

Site	Parameter	Periods	*df*	*t*	*P*
5	DO	1973-2019	1,438	8.48	**<0.001**
River Skerne	BOD	1973-2008	1,400	-14.71	**<0.001**
	Ammonia	1973-2019	1,452	-21.07	**<0.001**
	Nitrate	1973-2019	1,448	-3.77	**<0.001**
	Orthophosphate	1978-2019	1,311	-7.85	**<0.001**
	Zinc	1974-2004	1,179	-6.74	**<0.001**
	Lead	1989-2003	1,145	-2.42	**0.017**
6	DO	1976-2017	1,306	-0.62	0.539
Clow Beck	BOD	1976-2013	1,338	-8.36	**<0.001**
	Ammonia	1976-2017	1,377	-8.96	**<0.001**
	Nitrate	1976-2017	1,377	-4.10	**<0.001**
	Orthophosphate	1978-2017	1,276	-7.64	**<0.001**
	Zinc	1989-2013	1,247	-2.39	**0.017**
	Lead	1989-2003	1,102	-0.82	0.417
7	DO	1984-2019	1,242	1.44	0.152
Egglestone Abbey	BOD	1984-2006	1,225	-2.04	**0.043**
	Ammonia	1984-2019	1,308	-5.71	**<0.001**
	Nitrate	1984-2019	1,236	-3.38	**<0.001**
	Orthophosphate	1995-2019	1,244	-4.62	**<0.001**
	Zinc	1992-2019	1,231	-4.23	**<0.001**
	Lead	1992-2019	1,185	-3.54	**<0.001**

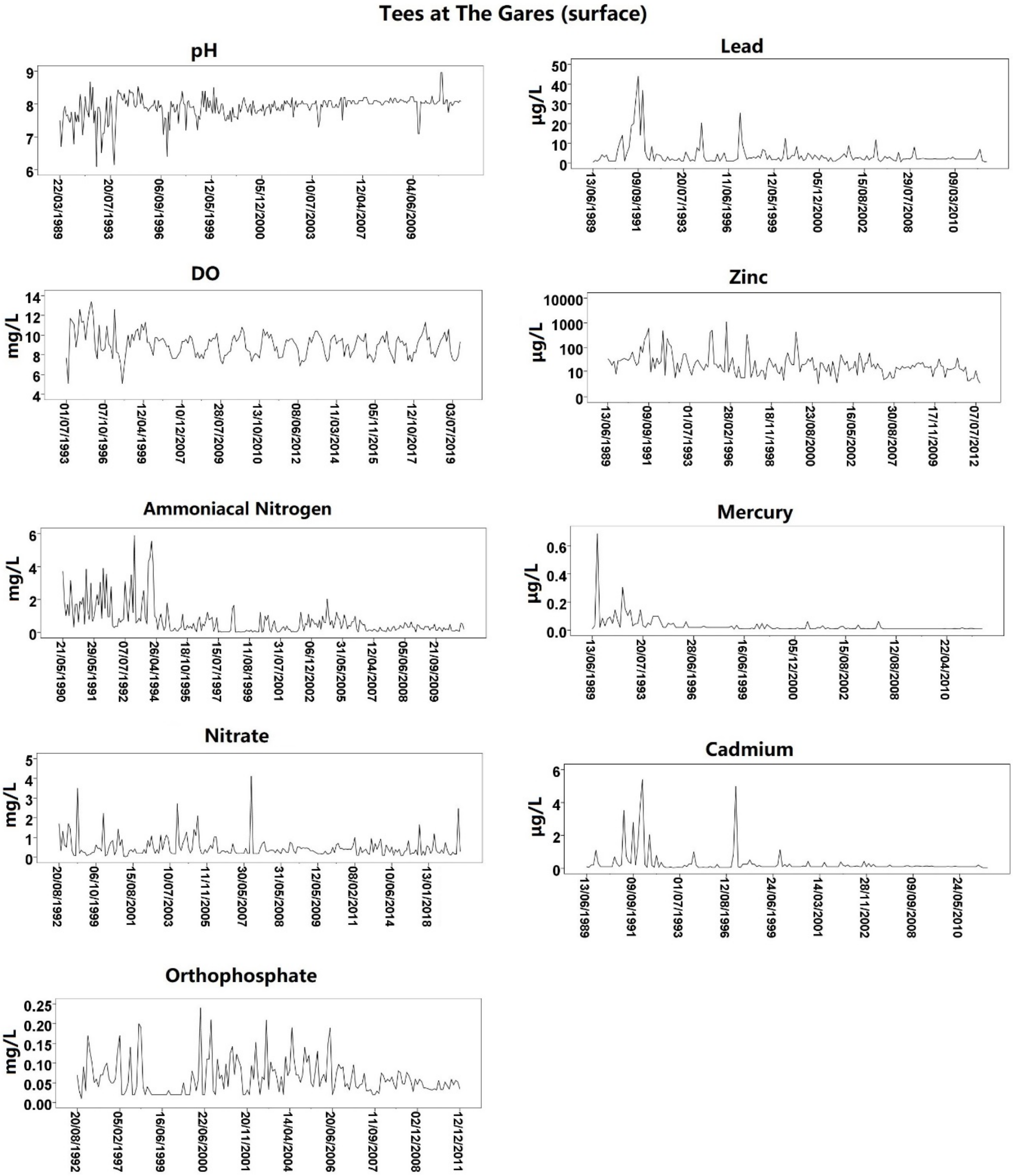


Figure 2.39 Key water quality parameters in the Tees at The Gares from 1989 to 2019. Note: zinc concentrations are on a log scale. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

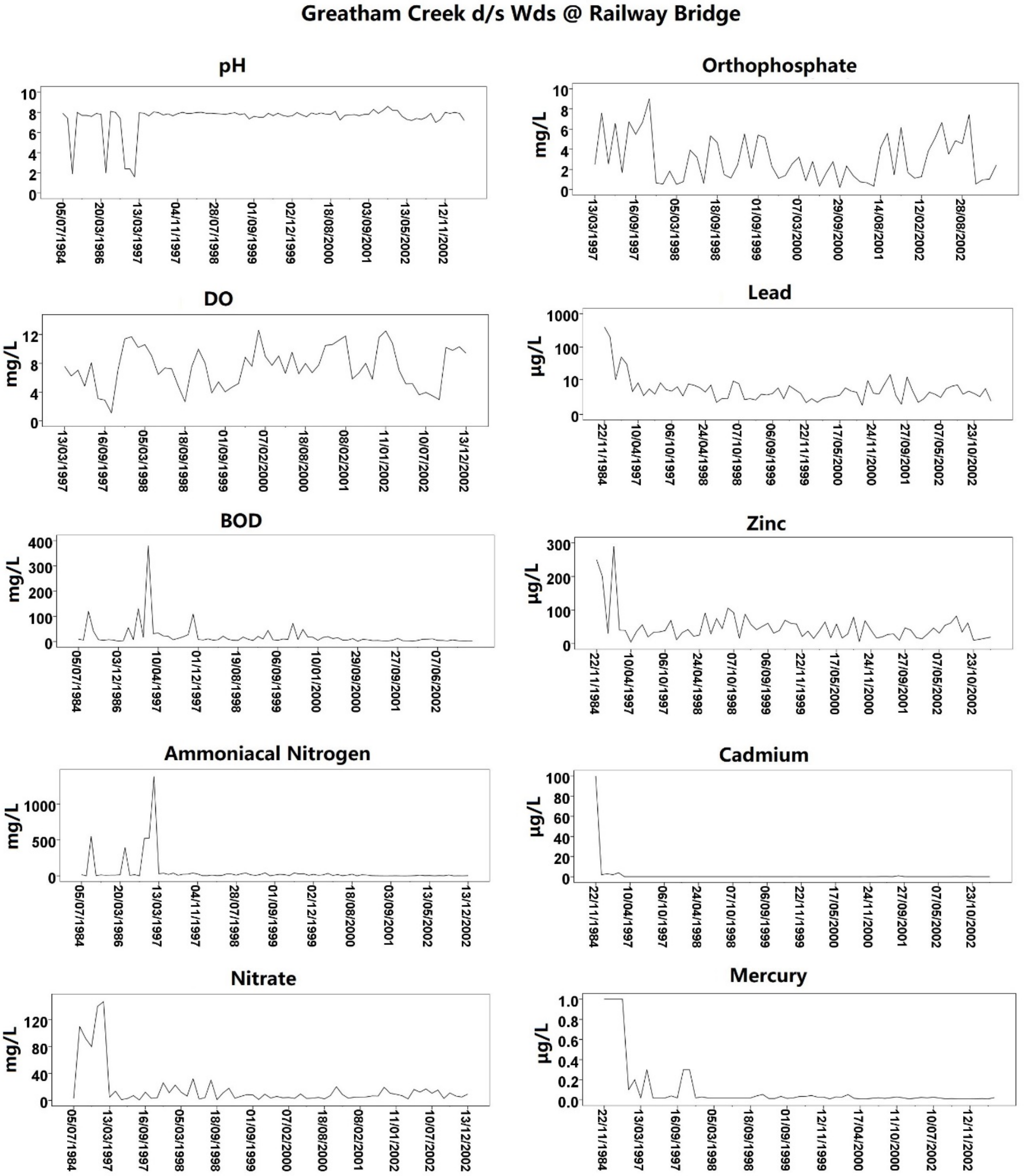


Figure 2.40 Key water quality parameters in the Greatham Creek d/s Wds @ Railway Bridge from 1984 to 2002. Note: lead concentrations are on a log scale. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

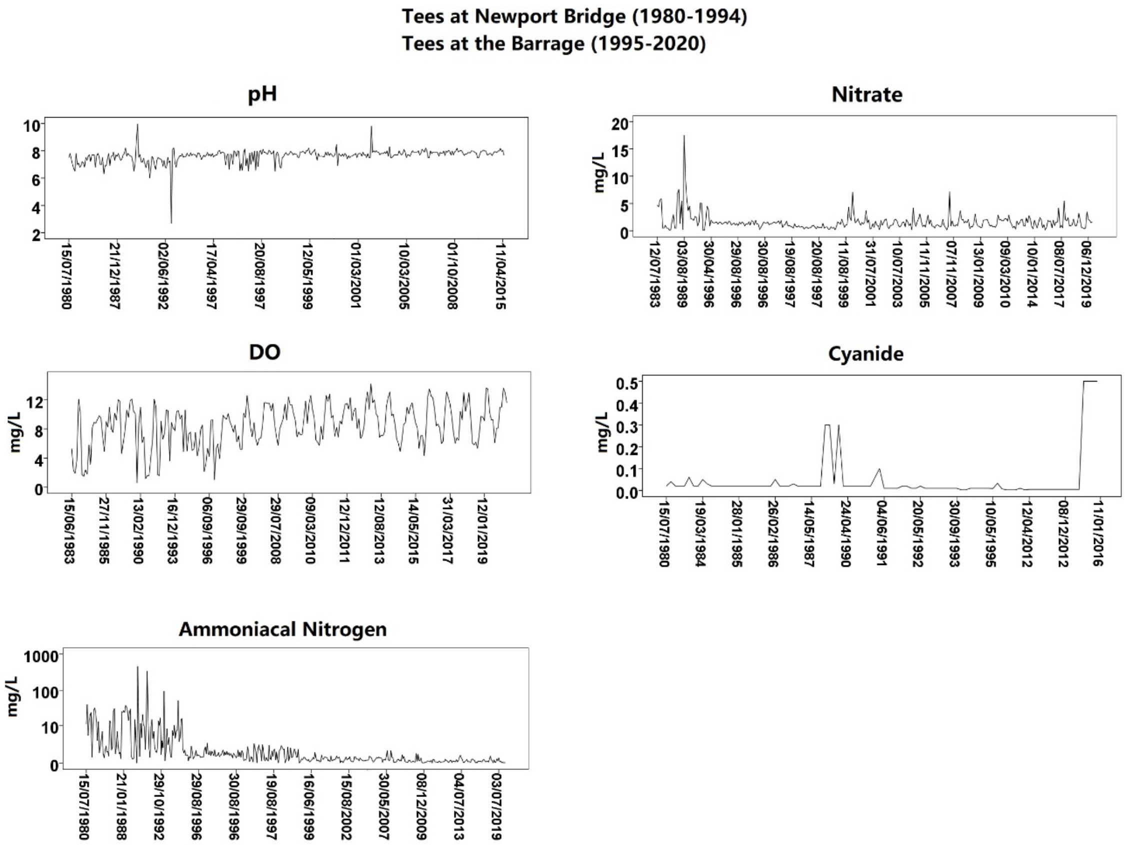


Figure 2.41 Key water quality parameters in the tidal reach (Tees at Newport Bridge from 1980 to 1994, Tees immediately downstream of the Barrage from 1995 to 2020). Note: ammonical nitrogen concentrations are on a log scale. Note the different timescales on the panels. The 1980 data were extracted from Ord (1988).

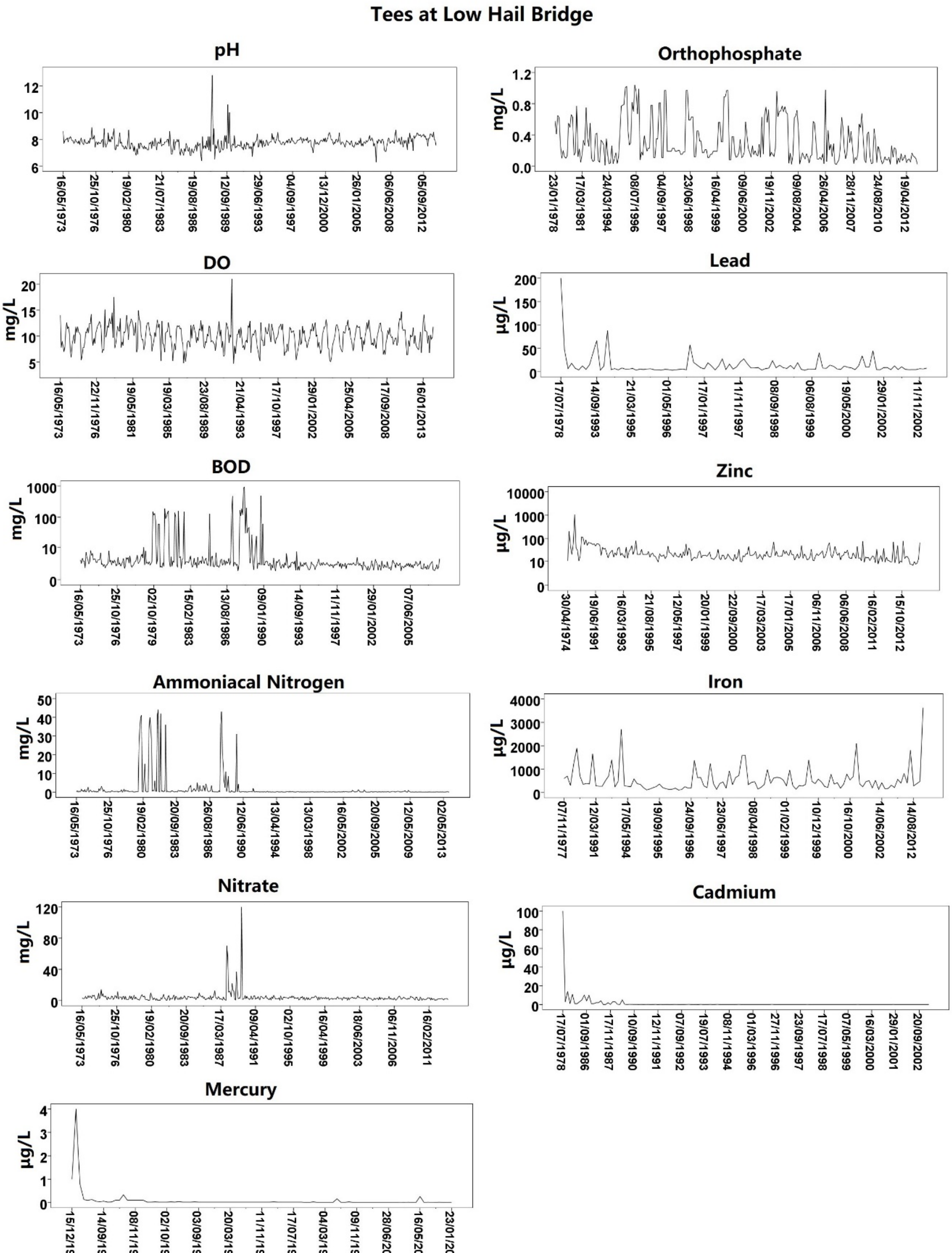


Figure 2.42 Key water quality parameters in the Tees at Low Hail Bridge from 1973 to 2019. Note: BOD and zinc concentrations are on log scales. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

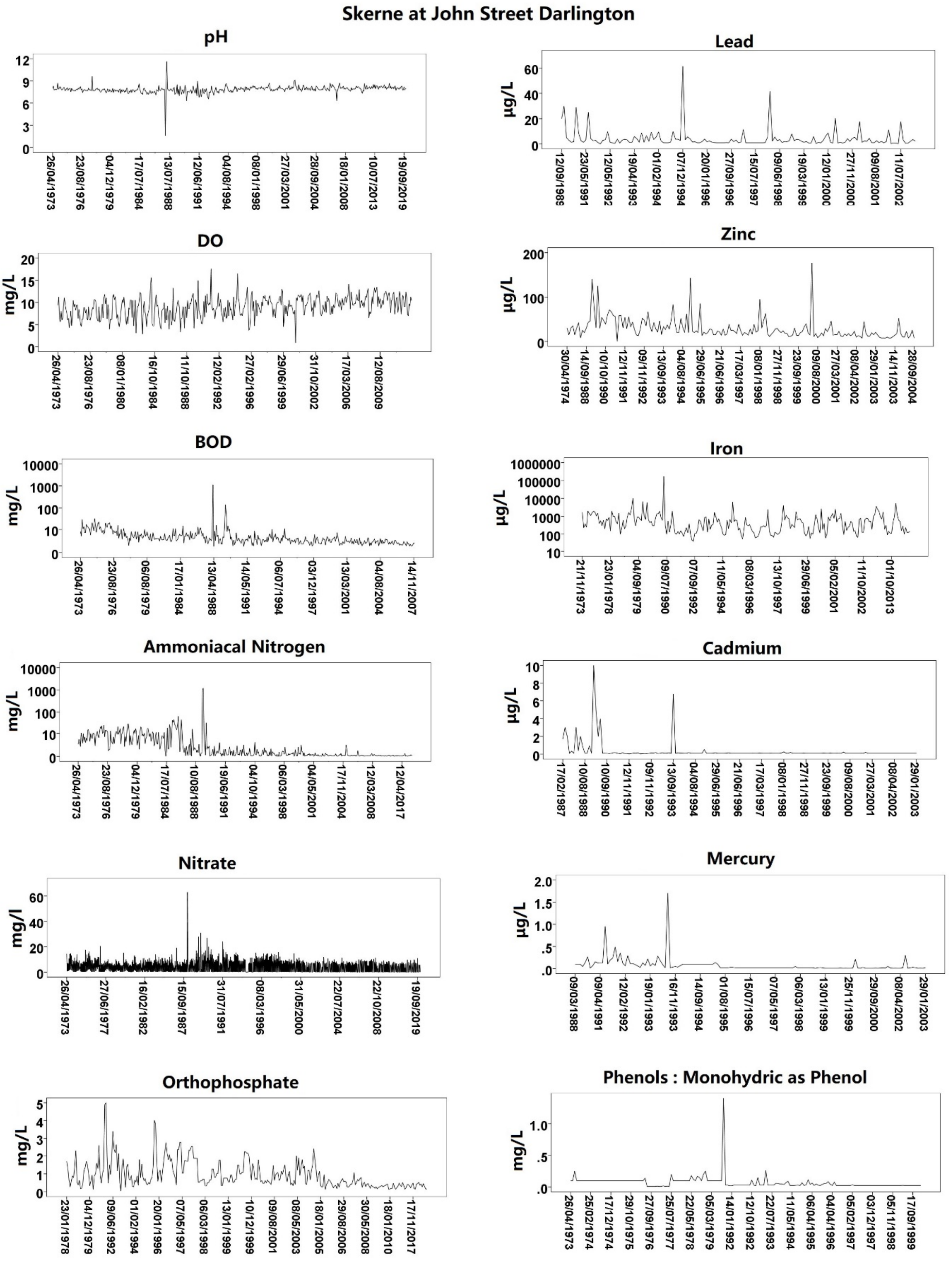


Figure 2.43 Key water quality parameters in the Skerne at John Street Darlington from 1973 to 2019. Note: BOD, ammoniacal nitrogen and iron concentrations are on log scales. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

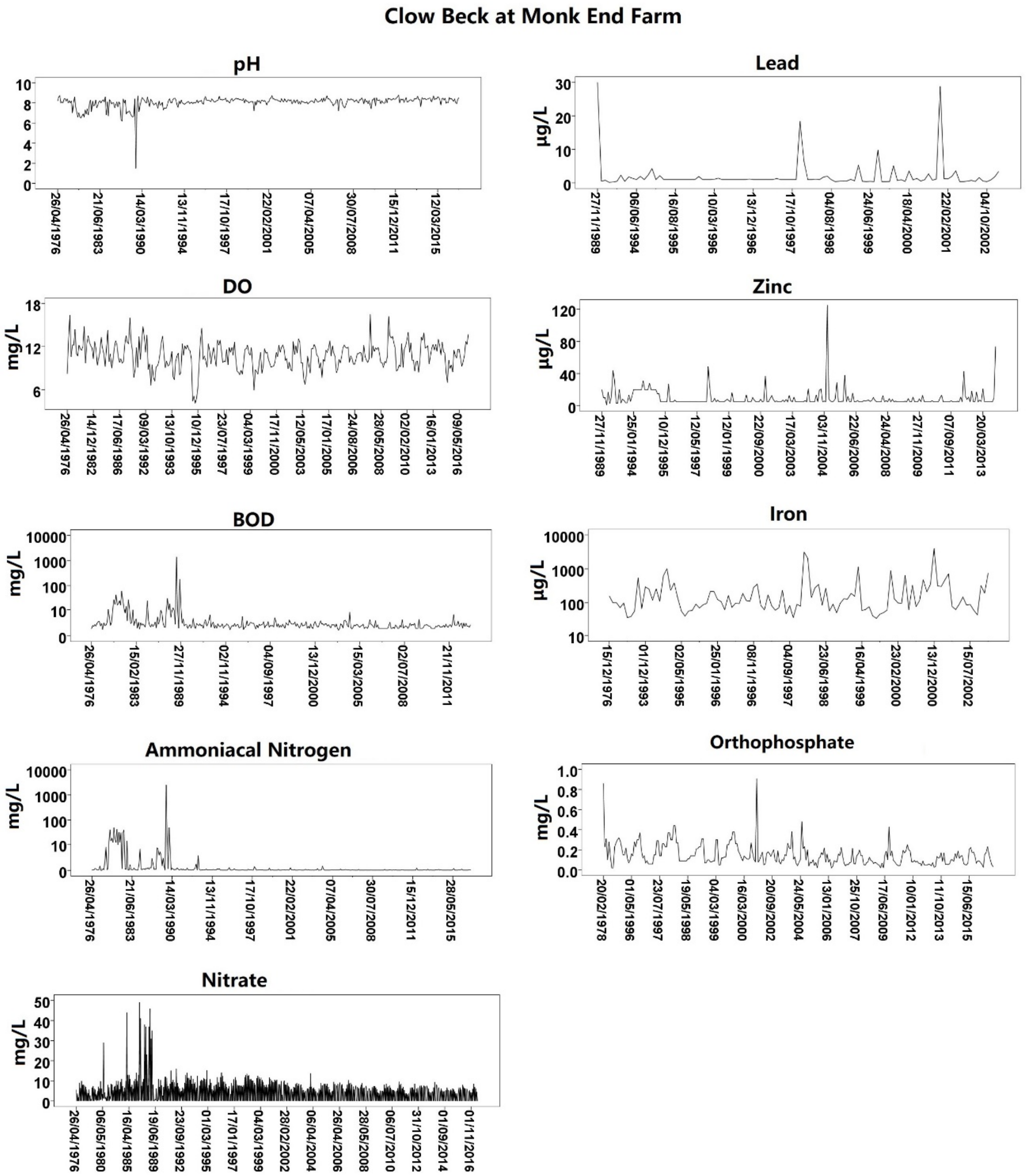


Figure 2.44 Key water quality parameters in the Clow Beck at Monk End Farm from 1976 to 2019. Note: BOD, ammoniacal nitrogen and iron concentrations are on log scales. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

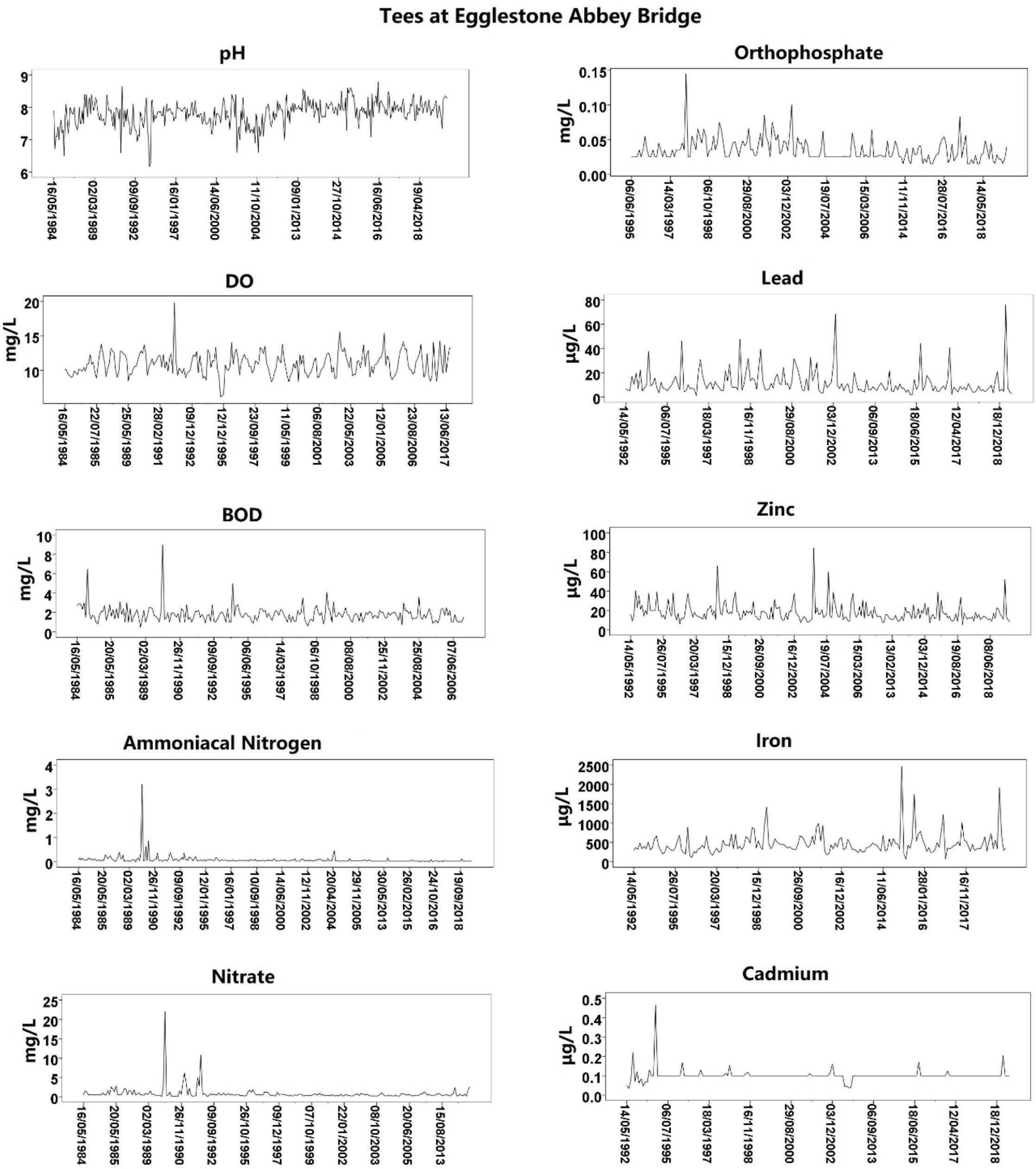


Figure 2.45 Key water quality parameters in the Tees at Egglestone Abbey Bridge from 1984 to 2019. All metal element concentrations presented are 'total' values (samples not filtered). Note the different timescales on the panels.

As shown in the analyses above, and similar to the Tyne and Wear, improvements in water quality have occurred in the Tees over the period 1973 to 2020, with decreases in determinands such as BOD and ammonia, characteristic or organic pollution, and decreases in heavy metals. Similar to the Tyne and Wear catchments, most sites were graded 'good' for chemical status in the Tees in both 2015 and 2016 in the EA's second WFD cycle (Table 2.11). However, after new rules and standards applied by the EA (Environment Agency, 2020d), all water bodies in the Tees catchment failed to achieve good chemical status in 2019 (Table 2.11).

No clear changes in the ecological status were found between 2015 and 2019 in the Tees, and the majority water bodies 74/87 (85.1%) failed to reach good ecological condition in 2019, with the greatest pressures coming from hydromorphological modification, pollution from wastewater, rural and urban pollution sources (Table 2.12).

Table 2.11 Ecological and chemical classification for surface waters in the Tees catchment in 2015, 2016 and 2019.

Tees catchment		Ecological status or potential					Chemical status	
Year	Number of water bodies	Bad	Poor	Moderate	Good	High	Fail	Good
2015	87	11	17	49	10	0	6	81
2016	87	7	19	50	10	1	7	80
2019	87	9	18	47	13	0	87	0

Table 2.12 Issues in the Tees catchment currently preventing waters reaching good status (second WFD cycle, 2015-2021) and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land manage-ment	Domestic General Public	Local and Central Gov	Mining and quarrying	Navi-gation	Urban and transport	Water Industry	Other	No sector respon-sible	Sector under investi-gation	Total
Changes to the natural flow and levels of water	1	-	-	-	-	-	7	-	-	1	9
Pollution from rural areas	73	-	-	-	-	-	-	-	-	-	73
Pollution from abandoned mines	-	-	-	11	-	-	-	-	-	-	11
Pollution from waste water	-	2	-	-	-	-	55	3	-	-	60
Physical modifications	18	1	24	-	15	17	8	3	-	3	89
Pollution from towns, cities and transport	-	6	1	-	-	6	34	-	-	-	47
Non-native invasive species	-	-	-	-	-	-	-	-	1	-	1

Several major river restoration projects have taken place in the Tees catchment since the 1990s, one of which was the River Skerne Restoration Project, which aimed to demonstrate rehabilitation of channel hydromorphology. The project was supported by European Commission LIFE funding with financial support from the Environment Agency (NRA), Darlington Borough Council, Northumbrian Water, English Nature and the Countryside Commission (Holmes and Nielsen, 1998). The project was managed by the River Restoration Project (the precursor to the River Restoration Centre) with significant additional partner management input from Darlington Borough Council and Northumbrian Water (Holmes and Nielsen, 1998). Restoration work started in July 1995 and was completed in September 1995. The restored section was a 2 km straightened reach on the outskirts of north-eastern Darlington (Vivash *et al.*, 1998). The project's aims were to install a coarse sediment riffle with some deflectors at the upstream end of the restoration reach, and to enhance flow variability and habitat diversity. In the downstream reach, four new meanders were created to cut across the old channel. Two backwaters were created, one located at the middle reach of the meander section, the other one located at the downstream end of the meanders. After restoration, the section has had an improved landscape with more natural habitat features. The channel length on the restoration reach increased 13% after the work was completed (River Restoration Centre, 1999). One year after restoration, the plant richness within the restored reach rapidly recovered, the species richness increased by 43% (including nine intentionally reintroduced plants) (River Restoration Centre, 1999). Before restoration, the river reach was dominated by resilient invertebrate species including water hoglouse (*Asellus aquaticus*), leeches and various water snail species. However, the invertebrate fauna had not improved two years after the restoration (River Restoration Centre, 1999). The fish species richness slightly increased at the restored reach two years after restoration, but this was suggested to be due to the newly installed fish pass further downstream at South Park in autumn 1996 (see section 2.3.3.6).

2.3.3.5 Recovery of the Tees fishes

From the 1930s until the 1980s, when the Tees began to recover, there was no salmon rod fishery there due to the severe pollution problems and resultant lack of anadromous salmonids. Reduction in industrial discharges to the estuary resulted in increases in dissolved oxygen levels (section 2.3.3.4), making the passage of anadromous salmonids possible (Environment Agency, 1999c). Between 1985 and 1996, the Tees was stocked with juvenile salmon from the Kielder hatchery. A total of 1.29 million fish were stocked at annual rates of up to 108,000 and at ages 0+ (72%) and 1+ (22%) (Environment Agency, 2009b). Salmon rod catches have increased on the Tees since 1982 (Figure 2.15). The

10-year average rod catch between 1982 and 1991 was 13.7 and it increased to 96.2 between 2008 and 2017. The annual catches reached a peak in 2008, when 267 salmon were caught. The declared sea trout rod catch has followed a pattern similar to salmon, starting to increase in 1995 and reached a peak of 143 in 2004. There has been a fishing pressure about 2,000 (salmon/sea trout) angler days per year since 1994 on the Tees (Environment Agency, 2009b), markedly less than on the Tyne and Wear. The catch per unit effort for salmon slowly increased between 1993 and 2008, then fluctuated between 2009 and 2017. For sea trout, the CPUE has been fairly stable since 1998. However, the Tees still has a much lower CPUE for both salmon and sea trout than for the Tyne and Wear (Figure 2.16). The difference of rod catches between the Tyne, Wear and Tees can be partly explained by variation in their specific fishery circumstances (e.g. regulations, locations, effort) as well as habitat quality and quantity (Environment Agency, 2008c), but the difference in CPUE is also reflective of the lower relative abundance of adult salmon and sea trout in the Tees and is reflected by electronic fish counts (Figure 2.18) one to two orders of magnitude lower in the Tees, even though these are partial counts of each river's population.

The recovery of anadromous salmonids in the Tees started late compared with the Wear and Tyne. It has been substantial since 1982, but with considerable fluctuations until the late 1990s (Environment Agency, 2009b), since when the improvement has halted, or even declined (Figures 2.15, 2.16, 2.18). The fluctuation during the early period appears to be common in the early years of salmon stock recovery in polluted rivers and probably reflects the initially inconsistent year to year water quality improvements along with the random variation when river stocks of salmon were low (Environment Agency, 2009b). However, since rod catch peaks in 2008 and 2012, the latter associated with an upstream Denil fish pass count of 1661, Tees fish counts have numbered only several hundred, although undoubtedly many fish are passing by other routes including the kayak slalom and the barrage gates.

In 1992, a fisheries survey (seine netting, electric fishing, including by boat) was undertaken in the middle to upper reaches of the main Tees between Croft and Middleton-in-Teesdale. In addition, surveys were also conducted on several tributaries. Dace was the dominant species in the lower sampling reach; dace, chub and brown trout were the dominant species in the middle reach between Croft and Winston; further upstream was dominated by trout, salmon and grayling (Owen *et al.*, 1993). No YoY salmon were caught during the survey and it was suggested that a large proportion of brown trout in the main River Tees were stocked by angling clubs (Owen *et al.*, 1993). The same study showed that Clow Beck, River Greta and Langley Beck held a good brown trout population. In contrast, no fish were caught in Billingham Beck and Lustrum Beck (Owen *et al.*, 1993). In 1993, a juvenile salmonid electro-fishing survey was conducted at 31 sites on the main river and 18 tributaries within the upper and middle Tees catchment from upstream Skerne-Tees confluence (National Rivers Authority, 1993b). Wild salmon were only recorded in Eggleston Burn and hatchery salmon were recorded at two main river sites and another six sites on tributaries (National Rivers Authority, 1993b). Comparing with salmon, trout were recorded at 29 surveyed sites, and 0+ trout were recorded at 24 sites (National Rivers Authority, 1993b).

To study the effects of stocking on Tees salmon recovery, 48% of the 1+ salmon were tagged before releasing to the Tees catchment (Jowitt and Russell, 1994). Among 102,794 tagged salmon, six were captured by Tees rod fisheries by 1994, a total of 411 recaptures were reported by other rod and net fisheries by 1995 (Shelley, 1994). The study suggested that Tees salmon stocking had insignificant benefits, and due to scarcity of recapture data, the contribution of restocking to the Tees salmon recovery cannot be reliably quantified (Shelley, 1994).

Besides salmon and sea trout, approximately 5,200 dace and 2,500 barbel were stocked in the Tees between October 1991 and May 1992 (Owen et al., 1993). In 1992 and 1993, approximately 24,000 brown trout and 1,000 dace were stocked in the River Skerne (Jenkins, 1994). Stocking with cyprinid and salmonid (brown trout and grayling) fishes has been continued since then, with an aim of supporting the rehabilitation of the Tees fish population, concentrating on sites where pollution incidents had occurred but where water quality had recovered and habitat was suitable. In recent years EA bylaws across England have prevented angling clubs stocking non-sterile brown trout unless they are from local broodstock and approved by the EA. The result of this on the Tees (as well as Tyne and Wear) is that stocking of brown trout has almost stopped, benefitting wild Tees trout stock integrity, but reducing the size of non-anadromous brown trout caught. There has also been some interest from local landowners for a Tees salmon hatchery, but this has effectively been resisted by the EA (M. Lucas, pers. comm.) based upon contemporary evidence of the effects of salmon stocking (Hansen, 2002; Miller *et al.*, 2004).

The recovery of other fish species also been recorded in the Tees catchment. River lamprey were recorded in 1995 during routine fish population surveys of the middle reaches of the Tees (Environment Agency, 1999c). Adult sea lamprey were recorded below the Tees barrage in 1996, 1997 and 1998 at the Denil fish pass (Environment Agency, 1999c), but free passage past the barrage has not been demonstrated for this species. After the construction of the Tees Barrage, roach, bream (*Abramis brama*) and chub fry numbers increased in the impounded section (Welton *et al.*, 1999b). Older dace, chub and roach densities also significantly increased in the impounded reach post-barrage (Welton *et al.*, 1999b), reflected in persistent increases in anglers' catches (M. Lucas, pers. comm.).

2.3.3.6 Changes of fish communities in the River Tees

Eight sites were chosen for study of long-term fish community change in the Tees catchment. One site was located in the main river and rest sites located in tributaries (Figure 2.46).

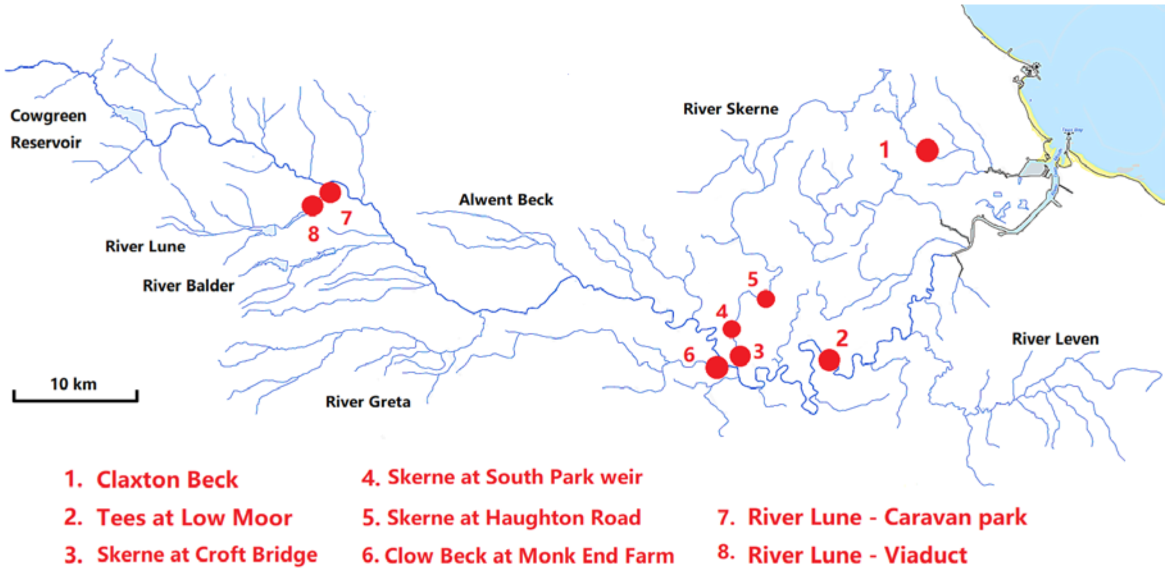


Figure 2.46 Tees catchment and location of long-term fish monitoring sites for which data is presented below.

The Newton Hanzard site of Claxton Beck (= 'North Burn'), is located 16 km upstream of the Tees estuary-Greatham Creek confluence. It used to be an NRA trout stocking site, at which approximately 10,000 trout fry were released in March 1997. Trout were present in low densities between 1998 and 2004 (0.6 to 2.8 per 100 m^2), but were not caught in 2011, suggesting they died out. In 1995, very few bullhead were caught (0.2 per 100 m^2) at the site, but abundance steadily increased to 26.5 per 100 m^2 in 2011 (Figure 2.47). Abundance of stone loach and three-spined stickleback both steadily increased between 1998 and 2011.

In the main river Tees, the Low Moor site (located 42.2 km upstream of Tees mouth) was chosen as a major coarse fish spawning site, unaffected by the Tees barrage impoundment (Welton *et al.*, 1999b). It was surveyed by seine netting of fry/small juveniles between 1993 and 2006. Pre-barrage, cyprinid fry were dominated by dace, chub and roach and this has not changed post-barrage (Welton *et al.*, 1999b). Eight fish species were caught during the netting surveys; minnow, dace and chub were present in high abundance during most sampling periods (Figure 2.48). In 2000, the roach fry abundance increased more than ten times compared with previous year. No salmonid fish were caught during the netting surveys. Since 2002, the EA also applied single-pass electrofishing on this site, with a total of 21 species caught up to 2019 (Figure 2.49). Tench (*Tinca tinca*), bream, rudd (*Scardinius erythrophthalmus*) and river lamprey were excluded from Figure 2.49 due to their low abundance (less than 0.01% of total captured numbers). Between 2002 and 2006, the community was mostly dominated by common minnow (0.8 to 12.8 per 100 m^2). Since 2009, the dominant species has shifted to bullhead (0.8 to 12.8 per 100 m^2). Stone loach (0 to 12.8 per 100 m^2), dace (0.1 to 3.5 per 100 m^2) and roach (0 to 3.4 per 100 m^2) were present at a slightly higher abundance compared with remaining species. Brown trout and Atlantic salmon were very rare (0 to 0.1 per 100 m^2). The overall fish community remained stable between 2002 and 2019.

In the River Skerne, three sites were studied. A total of 17 species were caught in three sites between 1991 and 2016 (Jenkins, 1992, 1994, 1998, 2000). Tench and mirror carp (*Cyprinus carpio carpio*) were excluded from Figure 2.50 because it was suspected that these two species were illegally introduced to the river from nearby fishing ponds (Jenkins, 1998). The Croft Bridge site is approximately 200 m upstream of the Skerne-Tees confluence. The fish community mainly consisted of minnow (minimum density, 1.7 to 171.2 per 100 m^2), dace (1.9 to 76.2 per 100 m^2), roach (0.1 to 10.8 per 100 m^2) and chub (0.2 to 8.8 per 100 m^2) between 1991 and 2004, other medium-large species were present in low abundance (Figure 2.50). Moderate abundance of eel was noticed at this site (0 to 17.1 per 100 m^2). The South Park weir site is located just downstream of Darlington, and is located immediately downstream of a stepped weir, where a pool and traverse type fish pass was constructed in 1996 (Jenkins, 1998). High abundance of eel (83.3 to 781.3 per 100 m^2) was recorded at this site in 1997 and 2000 (Jenkins, 1998). Since 2004, the dominant species has been minnow (5.2 to 8.3 per 100 m^2 – not quantified prior to 1997). Since 2004 at South Park site, the fish community had become less diverse and most fish species had declined in abundance. Similar trend was also observed at the Croft Bridge site since 2010. The Haughton Road site is located at the upstream end of the River Skerne restoration project reach (Jenkins, 1998). In 1991, no fish were caught at this site apart from three-spined stickleback (not displayed on the figure). In 1994 and 1997, a few brown trout, probably of stocked origin, were caught at this site (Jenkins, 1994, 1998). Dace, chub and roach were first recorded at this site in 1997. It was suggested that these species recolonized this site by using the South Park fish pass. However, these newly recolonized species may have been eliminated during a pollution incident on 14 November 1997 (River Restoration Centre, 1999). The species diversity at this site has increased a lot since 1997, but the overall fish abundance was still low during most sampling periods, except for high abundance of minnow (64.9 per 100 m^2) recorded in 2010 (abundance of 'minor' species not recorded before 1997). Overall, the fish community in the River Skerne recovered markedly in the 1990s, but has stalled since 2000.

Clow Beck is located in the middle Tees catchment. The Monk End Farm site, located immediately upstream of Clow Beck-Tees confluence, was electric-fished by the NRA in May 1992, then by the EA since 1995 (single pass on some occasions, multipass on other occasions). In the 1992 survey, abundant stone loach, bullhead and minnow were found, but numbers of these three species were not recorded during the survey (Owen *et al.*, 1993). A density of 18.4 per 100 m^2 of brown trout and 16.6 per 100 m^2 of eel were recorded (1992 data not shown in Figure 2.51), while other species were present in very low abundance (Owen *et al.*, 1993). From 1995, a total of 16 species were caught (Figure 2.51). The dominant species varied between minnow and bullhead between 1995 and 2002, then temporally shifted to brown trout in 2003 and 2004. From 2007 to 2014, minnow and bullhead (both species ranged from 22.7 to 328.9 per 100 m^2) were co-dominant species. Cyprinids, especially dace, roach and chub are common at the site. Salmon were present in most years, at low densities. Since 2014, minnow and bullhead abundance decreased, and brown trout became the co-dominant species with them.

The River Lune, located in the upper Tees catchment, is characteristic of a cobble and boulder dominated upland tributary, once important as a salmon spawning tributary, prior to impoundment. Four sites were surveyed by the EA since 1995 (single pass on some occasions, multipass on other occasions), and fish community data from the two most downstream sites (0.6 km and 1.4 km upstream of the Lune-Tees confluence) are considered. Eight species were caught during the electro-fishing surveys including a non-native species rainbow trout (*Oncorhynchus mykiss*). In the 1980s and 1990s this species successfully reproduced at this site (T. Crisp & M. Lucas, pers. obs.), probably supported by escapee rainbow trout from the reservoirs upstream, where sport fishing occurs. The use of sterile stock rainbow trout in the reservoirs upstream, and the resultant loss of mature adults, seems to have led to the demise of the feral spawning rainbow trout population (Figure 2.52). The dominant species at the Caravan Park site varied between brown trout (1.0 to 9.5 per 100 m^2), bullhead (0.1 to 79.1 per 100 m^2), Atlantic salmon (0.7 to 11.0 per 100 m^2) and minnow (0.1 to 11.5 per 100 m^2) (Figure 2.52). The fish community was relatively stable between 1998 and 2019, although abundance has varied somewhat over time. The Viaduct site has been surveyed inconsistently. It was mostly dominated by brown trout between 1995 and 2019, present in low abundance (1.0 to 14.2

per 100 m^2). Bullhead abundance slightly increased at this site between 2006 and 2019 (3.6 to 35.7 per 100 m^2).

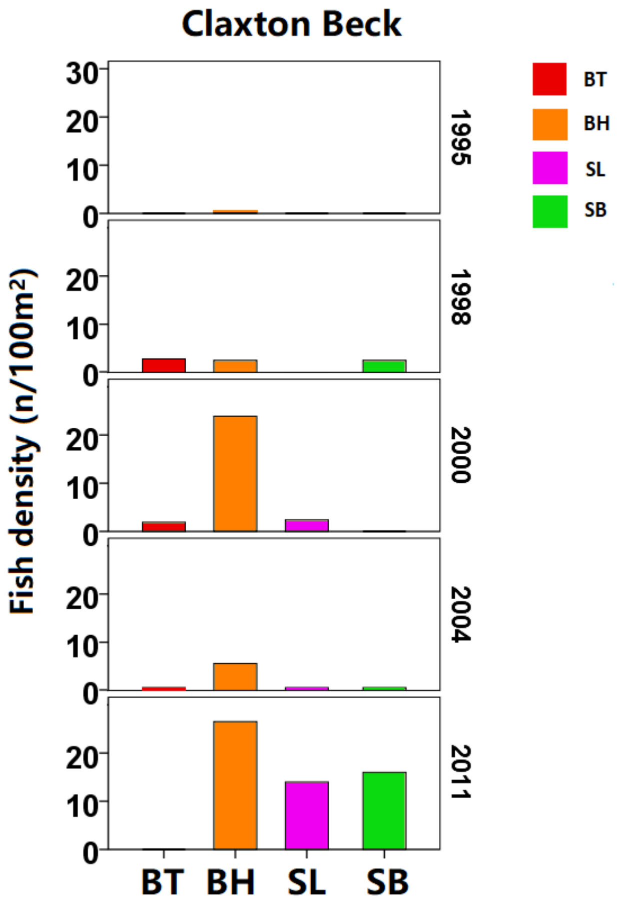


Figure 2.47 Long-term variation of estimated minimum fish density (single pass electric fishing) at Claxton Beck Newton Hanzard site between 1995 and 2011. BT: brown trout, BH: bullhead, SL: stone loach, SB: three-spined stickleback.

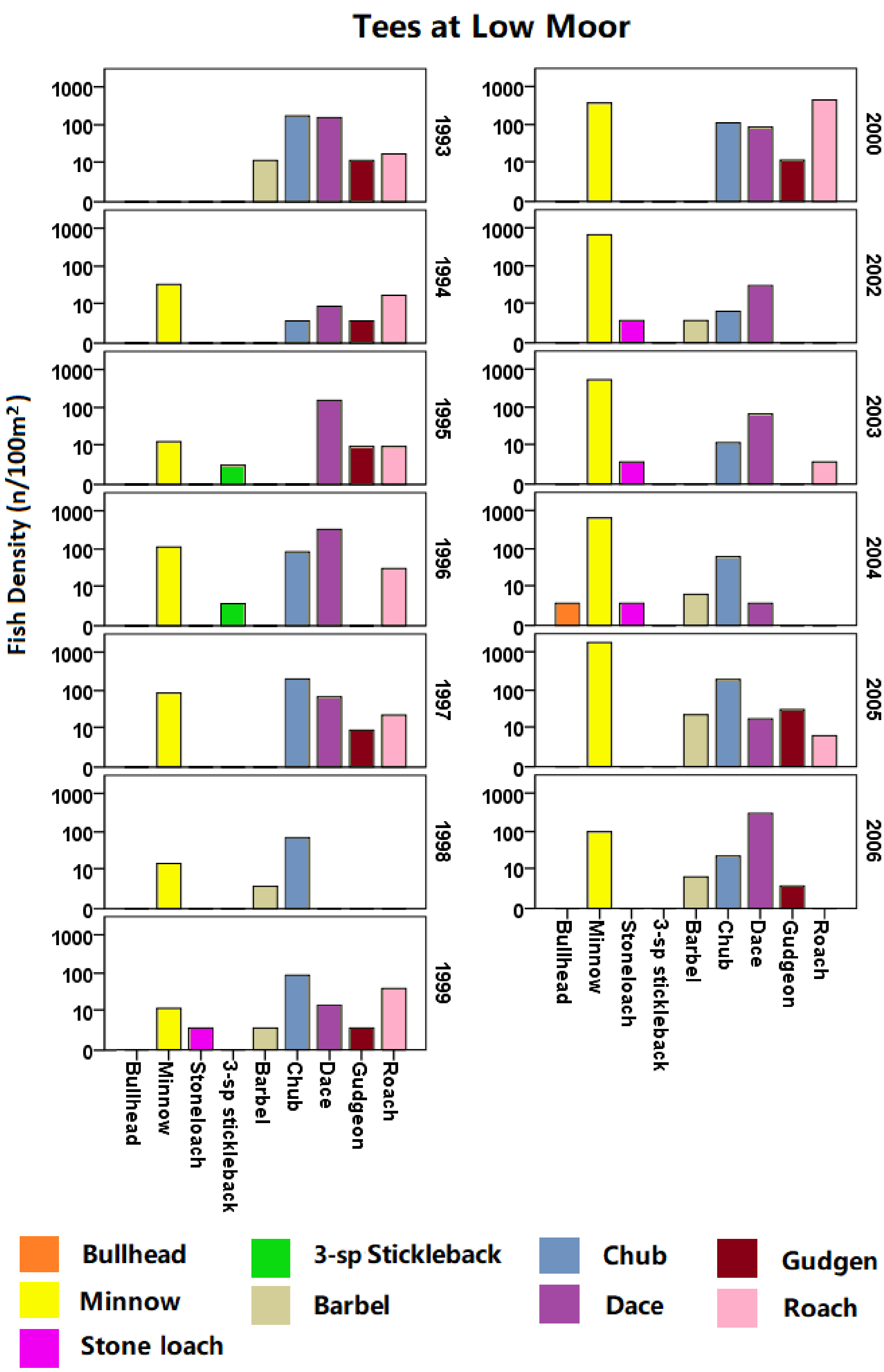


Figure 2.48 Long-term variation of minimum density of fish fry and small juveniles at River Tees Low Moor site between 1993 and 2006. Fish density was calculated by single catch seine netting method.

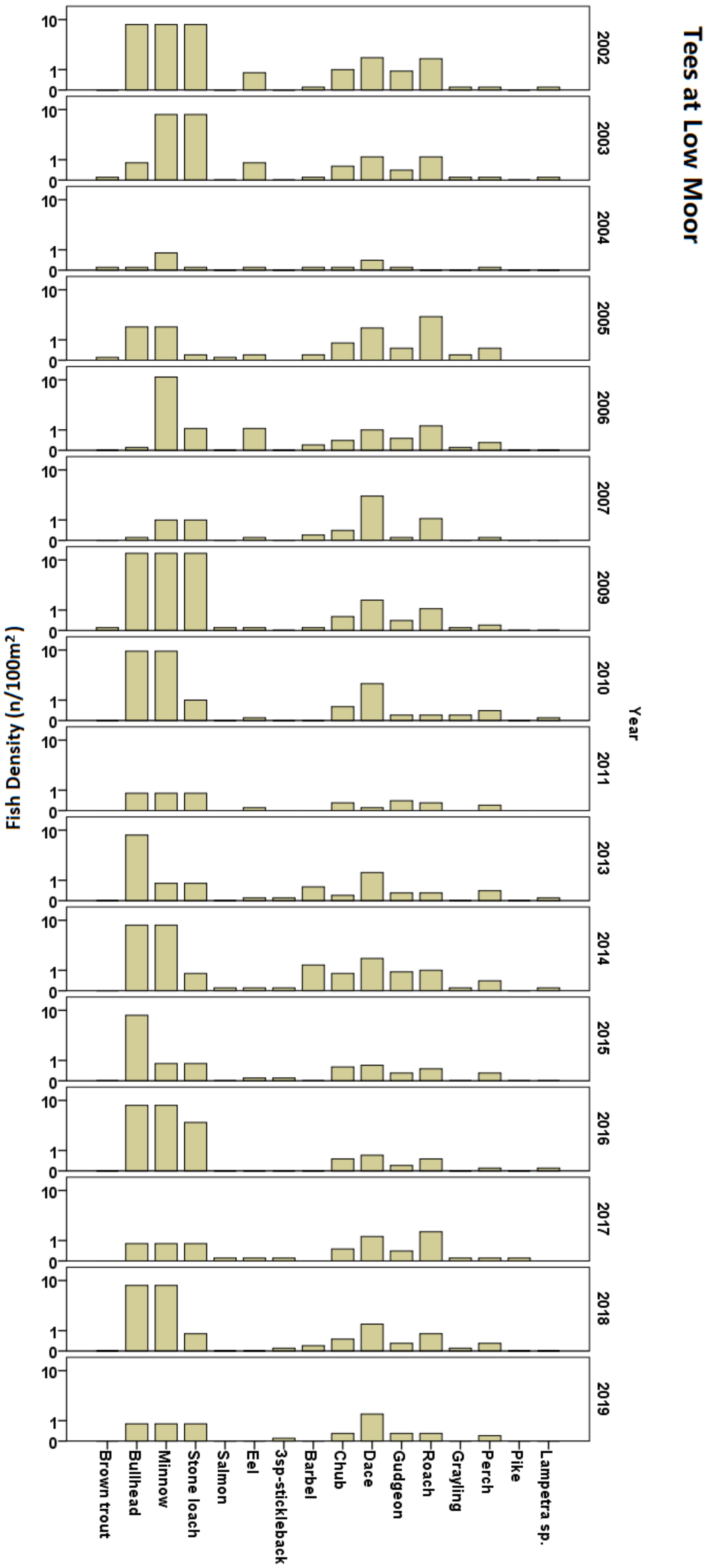


Figure 2.49 Long-term variation of estimated minimum fish density at River Tees Low Moor site between 2002 and 2019. Fish density was calculated by single pass electro-fishing method. Notice: abundance values are on log scales.

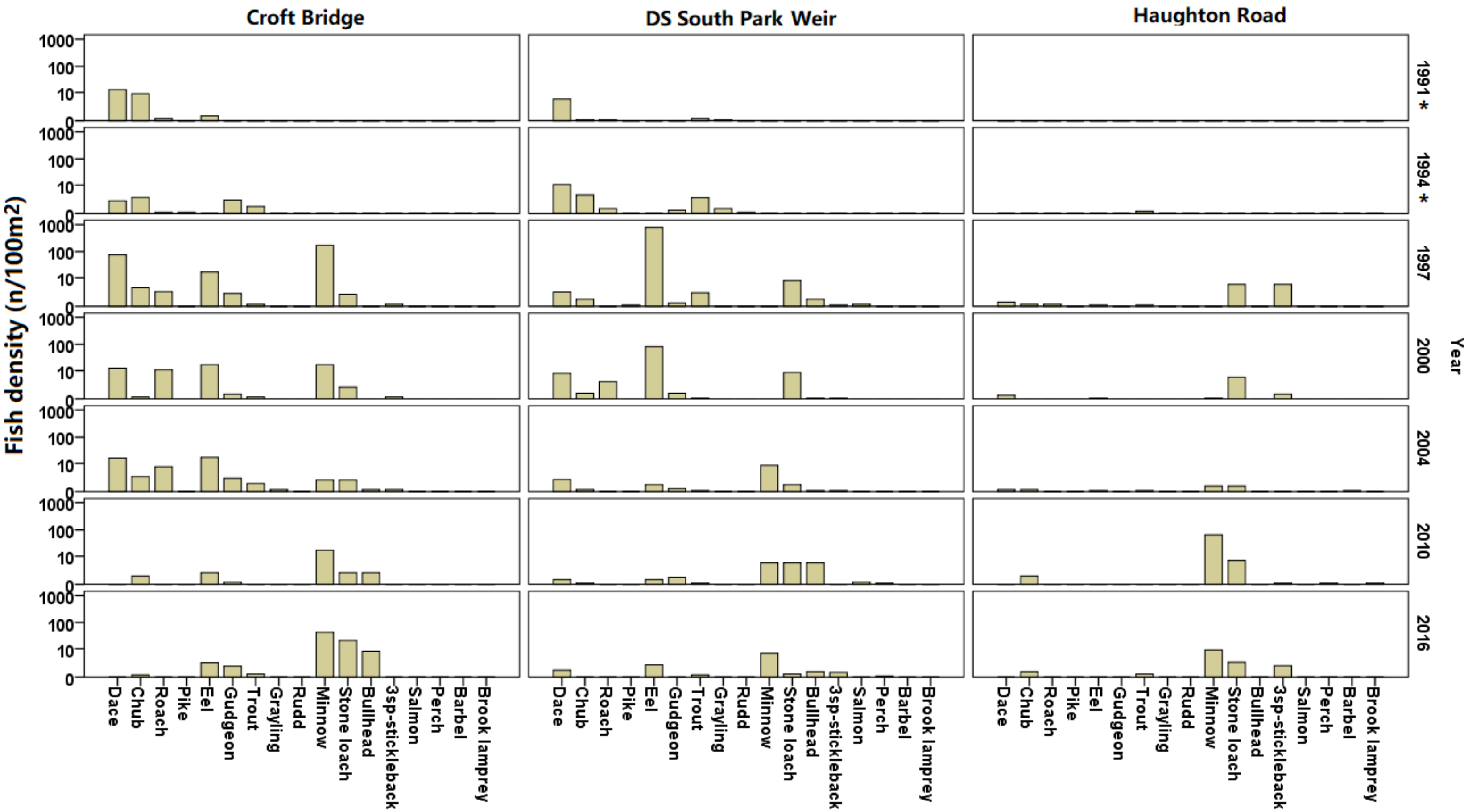


Figure 2.50 Long-term variation of estimated minimum fish density at River Skerne sites between 1991 and 2016. Fish density was calculated by single pass electro-fishing method. Note: abundance is on log scale. No fish were caught in 1991 at Haughton Road site. Minnow, stone loach, bullhead and stickleback may have been present but were not recorded or counted in 1991 and 1994 surveys for all sites.

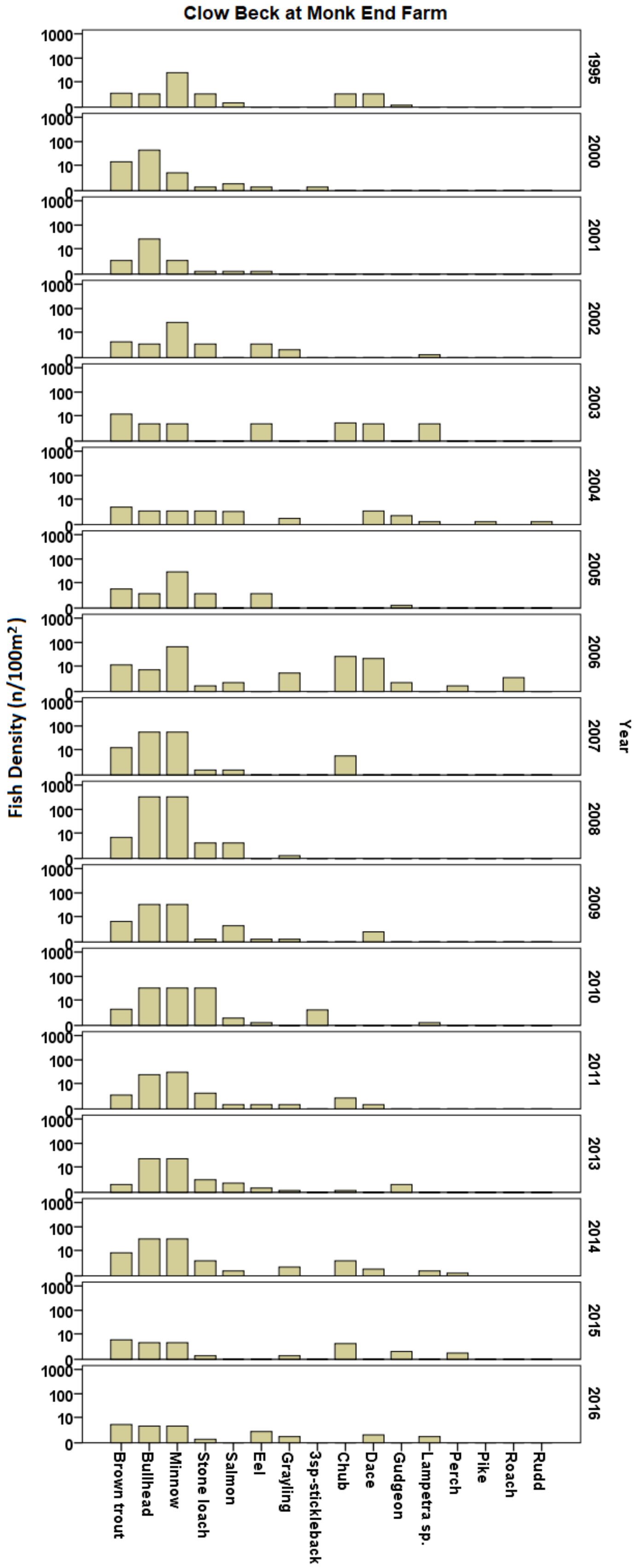


Figure 2.51 Long-term variation of estimated minimum fish density (1995, 2000, 2007-2016: single pass electric fishing; 2001-2006: first run data from three pass electric fishing at Clow Beck Monk End Farm site between 1995 and 2016 (log scale).

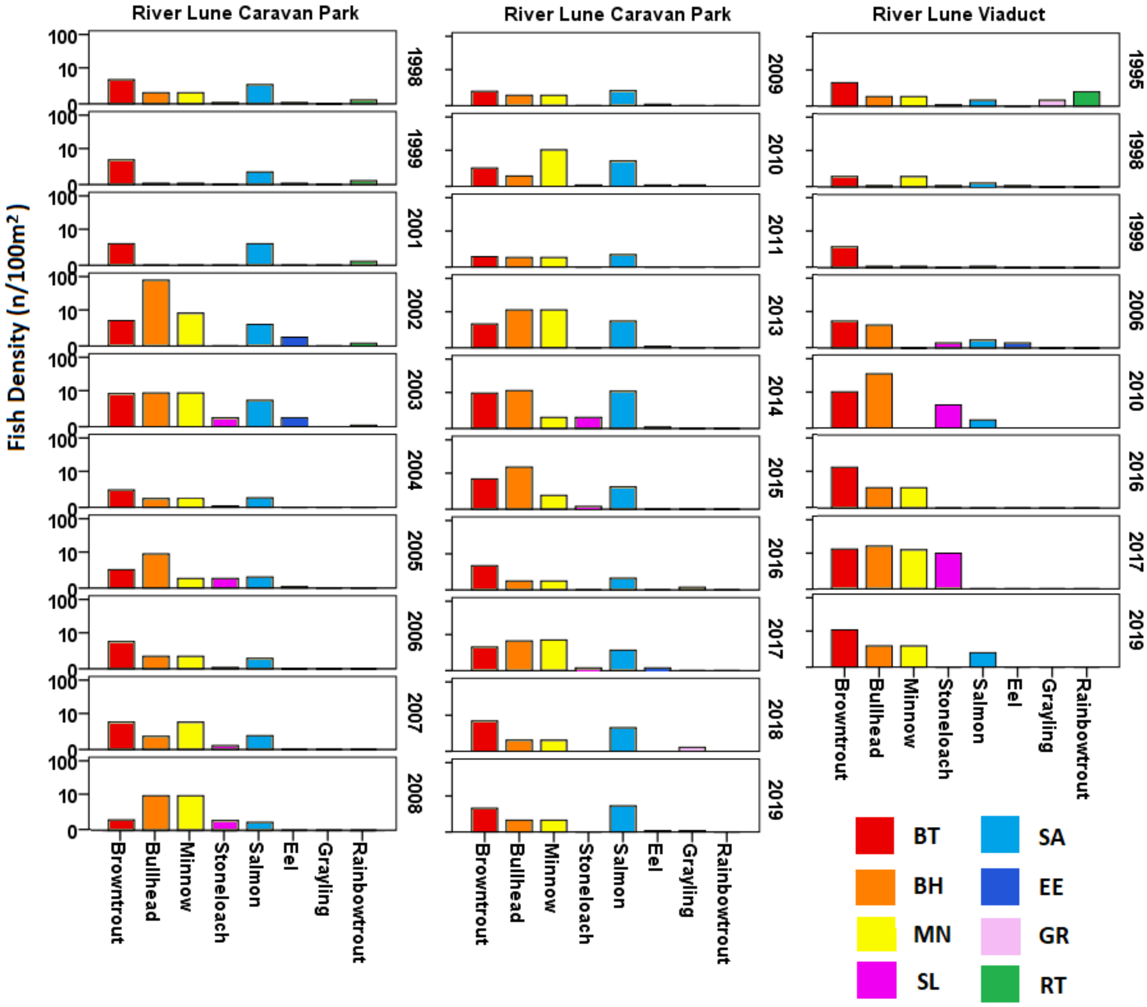


Figure 2.52 Long-term variation of estimated minimum fish density (1995, 1999, 2007-2019: single pass electric fishing; 1998: first run data from two pass electric fishing; 2001-2006: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 88.6%, salmon 96.8%) at River Lune sites between 1995 and 2019 (log scale).

2.4 Discussion

The study's key aims were met. Historical evidence concerning the nature of environmental impacts on the Tyne, Wear and Tees showed the timeline of decline and recovery of fish across the three catchments. Factors that led to the near extirpation of salmon and sea trout in these rivers have been identified (see below for further discussion). Information on changes in other fish species is mostly lacking until recent decades. Nevertheless, understanding the effects of these factors is a precondition for effective future restoration works. Mitigating these factors can help with the recovery of the threatened species, and this approach can be applied on other deteriorated river systems which have been affected by historic human activities (Langford *et al*., 2012).

Based on the historical evidence presented in Chapter 2 the main reasons for the decline and near extirpation of Atlantic salmon and the anadromous sea trout phenotype of *Salmo trutta* from North East post-industrial rivers can be strongly suggested to be due primarily to severe pollution, especially in the lower reaches, and river barriers. These impacts were probably exacerbated by fisheries and localized physical habitat modification. These conclusions support those elsewhere (Sheail, 2000; Archer *et al.*, 2003; Champion 2003) but are based on associative evidence sources from historical records. One possible future approach for examining temporal trends of pollutant impacts from the Tyne, Wear and Tees might come from microchemical analysis of archived salmon and trout scales if these exist (Morán et al., 2018). Certainly some scale collections from these rivers are available going back several decades, but whether any are available from much longer again (e.g. early 20th Century) is unknown.

In the Tyne catchment, the salmon and sea trout rod catch increased almost continuously from the 1960s to ~2010, but has declined somewhat since, reflected also in counter records. This may suggest that overall production capacity has reached its current limit and this is a natural expectation in a recovering river (Environment Agency, 2008c; Cefas *et al.*, 2019). Salmon and sea trout rod catch in the Wear catchment has shown a similar trend, paralleled by counter records. For salmon, there is concern that nationally, and more widely that salmon survival at sea has declined markedly in recent years (ICES, 2019). Thus the declines in salmon and sea trout abundance, although perhaps part of a natural cycle, need to be viewed with concern in light of the broader international evidence. Sensitive integrated management can still achieve rapid recovery of depleted salmon populations (Koed *et al.*, 2020). Although water quality, habitat quality and overall availability of spawning and juvenile habitat are greater in the Tyne than the Wear, substantial room for improvement in these, and access to good habitat remains in both rivers (sections 2.3.1.4, 2.3.2.4). For example, salmon exhibit rather little use of the River

Browney and Deerness (Wear catchment) after decades of recovery, despite the relatively good water quality, good physical habitat and accessibility of the Browney/Deerness. Is this a sign of stalled recovery (perhaps exaggerated by strong within-catchment philopatry) or a reflection that the Browney and Deerness are still suboptimal habitat for salmon?

Although banning intentional salmon netting along the Northumbrian and Yorkshire coasts now reduces pressure on NE England salmon, the increased quotas for sea trout (Environment Agency, 2020e) are a concern, especially since rod catches of Tyne and Wear sea trout, and Wear fish counts have decreased in the last decade. Enforced return of spring salmon by anglers and encouragement of catch and release for all other salmon by anglers (59% for Tyne and 51% for Wear in 2018; Environment Agency, 2018; Cefas *et al.*, 2019) measures can also aid the recovery of salmon stocks, so long as post-release survival to spawning is high. Currently there are no bag limits or method (fly, spinner, bait) restrictions on these rivers. The Tyne and Wear are among only 14 of 64 English and Welsh principal salmon rivers where spawning escapement is exceeding the calculated conservation limit (Environment Agency, 2018). Evidently, resurrection of the Tyne and Wear from open sewers without salmon runs, to the first and second best salmon rivers in England and Wales is a restoration achievement to be appreciated, but not underestimated.

For the River Tees, the recovery started about 30 years later compared with the Tyne and Wear catchments and the data suggest the river is still not achieving its current conservation limit for salmon egg deposition (Environment Agency, 2009a; Cefas *et al.*, 2019). The Tees Barrage, with currently inadequate passage, remains a major impediment to the recovery of anadromous salmonids in the Tees. The slower recovery of anadromous salmonids in the Tees and the relative impact of the barrage provides an interesting comparison of outcomes with the Tyne and Wear. There are many opinions among stakeholder groups on the causes of continued low Tees salmon numbers. Indeed some anglers are of the opinion that stocks are much higher than assumed and deliberately do not report their catches (which is illegal) in order to minimize ‘competing’ angler attention on the river (M. Lucas, pers. comm.).

A similar recovery for salmon, but not yet to the same degree as for the Tyne and Wear has occurred in the Clyde catchment in Scotland, where salmon and sea trout reappeared in the 1970s and started to recover in the late 1980s (Doughty and Gardiner, 2003). Apart from salmonids, river lamprey, sea lamprey and brook lamprey have been recorded in the Clyde in recent years (Hume, 2017). The main reason for fish recolonizing the Clyde was also (as for Tyne, Wear and Tees) the water quality improvement since the 1950s. Over the past 100 years, several sewage treatment works were built within the catchment (Doughty and Gardiner, 2003). Improved water quality in the major tributaries has subsequently allowed salmon return to the Clyde and re-established their population. In addition, limited stocking programme was also conducted in the catchment. Besides, in 1995 a fish pass was installed at the Blantyre Weir in the Clyde, which is a major barrier to fish migration (Doughty and Gardiner, 2003) and numerous other obstacles have been removed or fitted with fish passes. In recent years, barrier surveys conducted by the Clyde River Foundation have identified a number of barriers, providing valuable information for future connectivity restoration works (McColl *et al.*, 2009).

In the Mersey catchment, after water quality legislation referred to in Chapter 1 was introduced, water quality began to improve in the 1970s (Ikediashi *et al.*, 2012). More than £1 billion had been spent on controlling and cleaning the effluents to the river by the start of the 21st Century (Mawle and Milner, 2003). As a result of the improving water quality, salmon began entering the river Mersey in the early 1990s (Ikediashi *et al.*, 2012). Between 2001 and 2011, 158 untagged adult salmon were caught at Woolston weir by the Environment Agency (Ikediashi *et al.*, 2012). A tracking and genetics study has shown both stray and native salmon returned to the Mersey system, and successfully arrived upstream to potential spawning areas (Billington, 2012; Ikediashi *et al.*, 2012). Similar to the Tees catchment, the river is currently in the early stage of an on-going process of natural recolonization, following the substantial improvement in the overall river environment (Billington, 2012; Ikediashi *et al.*, 2012).

However, not every river has had a successful recovery of its salmon stocks after the restoration. The failure, to date, to restore wild salmon to the Rhine was described extensively in the Introduction to this chapter. In the River Thames, since the 1970s, water quality improved dramatically and salmon reappeared since 1974 (Griffiths *et al.*, 2011). In 1979, the Salmon Rehabilitation Scheme was established, in which Thames Salmon Trust released stocked salmon (mostly from Scottish hatcheries) into a variety of Thames tributaries (Mawle and Milner, 2003; Griffiths *et al.*, 2011). From 1979 to 2010, a total of 4.4 million juvenile salmon were released in the catchment (Griffiths *et al.*, 2011). Since then, the adult salmon numbers recorded in the river gradually increased, and reached a peak of 338 in 1993 (Griffiths *et al.*, 2011). Meanwhile, a fish pass construction programme was carried out, in which 36 weirs were installed with fish passes between the tidal limit and spawning habitat on the tributary River Kennet (Griffiths *et al.*, 2011). However, since 1997 the numbers of adult salmon recorded in the Thames significantly decreased, and no salmon were captured in 2005 (Griffiths *et al.*, 2011). Between 1998 and 2017, barely any salmon and sea trout were caught by rod in the Thames (Environment Agency, 2013a, 2019). Although lots of effort was spent improving river habitat and connectivity in the River Kennet, the Thames catchment still cannot hold a self-sustaining population of salmon. Conditions in the Thames in recent years are considered to be the reason that prevented fish from ascending the Thames (Griffiths *et al.*, 2011). Between 1989 and 2006, the number of storm sewage releases significantly increased, large volumes of high BOD effluents were released into the tidal reach of Thames (Griffiths *et al.*, 2011). All these factors potentially reduced the dissolved oxygen level and created a barrier to fish migration. Furthermore the Thames is in south east England in an area suffering the twin impacts of climate change and increasing human population density through pressures on water abstraction and water warming. By contrast, the northeast English rivers are currently much better buffered from these effects.

Although the recovery of salmon and sea trout was observed in the Tyne and Wear catchments, these catchments still face great challenges of river management for much fuller restoration. The WFD has been a strong driver of this in recent decades and in the short term the UK will adopt a path of environmental improvement paralleling WFD following Brexit. Only a small proportion of tributaries in these catchments have achieved good ecological status and all waterbodies failed the chemical status in 2019, after new substances were added to the chemical quality assessment and new standards were introduced, many persistent organic pollutants in English rivers were found at levels exceeding the WFD environmental standards (Environment Agency, 2020d). The long-term impacts of many of these are as yet not fully understood (Johnson *et al.*, 2013; Alharbi *et al.*, 2018). These pollutants continue to be dispersed in the environment, polluting rivers and hindering the recovery of freshwater ecosystems.

Evidence from the fish community survey records across the Wear and Tees shows that some tributaries have been increasingly recolonized by a range of fish species as they have become cleaner and habitat has improved, for example in the Cong Burn and Browney of the Wear and the River Skerne of the Tees. However, in several cases the recoveries appear to have stalled somewhat, for example in the Skerne in Darlington. This may reflect continued poor habitat over large areas, and periodically variable water quality causing poor recruitment. Although since the early 1990s there has been a trend in recovery of the fish communities in formerly nearly fishless streams of the Wear and Tees, the lack of data prior to 1991, and inconsistent sampling since then makes this difficult to quantify.

In the Tyne catchment, physical modifications such as instream barriers and channelization are the main reasons for not achieving good ecological status, followed by pollution from abandoned mines, and diffuse pollution from rural areas (Environment Agency, 2020a). In the Wear catchment, the most recent investigation by the EA has shown point and diffuse source pollution from metal mines within the upper catchment are the main reason for environmental status not achieving good status in some tributaries (Environment Agency, 2017c). Ironically this may not be particularly hindering the restoration of natural fish communities in the Wear catchment (at least at current background metal concentrations); instead river barriers, fine sediment and nutrient enrichment are likely to be more problematic. The water industry is most responsible for pollution from waste water, mainly from point source discharges from sewage treatment works and intermittent discharges from combined sewer overflows and diffuse pollution from wrong connections (Environment Agency, 2017c, 2020a). In rural areas, diffuse pollution from poor land management is contributing to silt entering watercourses and contributing to phosphorus and ammonia levels (Environment Agency, 2017c, 2020a).

Besides pollution, physical modifications are another reason for not achieving good status (Environment Agency, 2020a). River channel modifications such as urbanisation, mining infrastructure and flood defences formed the majority of physical modifications, mainly concentrated around Chester-le-Street and Durham (Environment Agency, 2017c). The impacts of impoundment reservoirs also need to be addressed across all three catchments. In the Tees catchment, physical modifications like instream barriers and channelization are also the main reasons for not achieving good ecological status, followed by pollution from rural areas, pollution from waste water and pollution from towns, cities and transport. The Tees Barrage is still considered to be one of the major obstructions to fish passage on the Tees, and it is unlikely to be removed in the near future. In addition, predation of fish by seals at the barrage has become another issue that has arisen in recent years. So, it is suggested that upgrades on fish passage could be done, to improve the attraction rate and facilitate more fish to pass quickly through the barrage during migration periods. Crucially fish passage at the Tees Barrage needs to accommodate all migratory species and in upstream and downstream directions (Silva *et al.*, 2018), whereas currently upstream passage of salmonids is only being emphasized.

Future works should be mainly focused on easing both point source and diffuse pollution, and restoring both physical habitat and connectivity on the Tyne, Wear and Tees catchments. These works are important components of catchment management. Improving the sewerage system to enable improvements in water quality, mitigating the impacts of metal and coal mine pollution, and reducing the impacts of urban diffuse pollution (Environment Agency, 2017c) continue to be key. When good habitat condition is guaranteed for expected native fish communities (and other biota), the next key stage will be to identify undiscovered in-stream barriers in tributaries and create free passage for fish to access the previously inaccessible habitat. Apart from salmon and sea trout, improved conservation of other aquatic species such as European eel, river lamprey and white-clawed crayfish is also needed. And when habitat becomes suitable, reintroduction of extirpated species such as smelt may be considered.

In the meantime, it is apparent how crucial reliable quantitative recording and safe archiving of fish community data is at a network of sites, and this needs to be a priority for the future. The lack of stream fish community data that could be recovered for before 1991 is evidence of the extent of this problem. The same is also true for environmental data such as water quality parameters. It is evident that many data series available from the Environment Agency and its predecessors for the Tyne, Wear and Tees were of short duration and suffered problems of changes in location and method that made assessment of change difficult. For most fish species, accessible quantitative data are available only since 1990. Even then, many fish species – loach, bullhead, minnow, eel, lampreys, sticklebacks were not properly recorded until 2004 onwards, a direct result of WFD monitoring. The success of fish community restoration can only be evaluated if the data exist to do so.

2.5 Conclusions

This study reconstructed the timeline of the decline and partial recovery of the rivers Tyne, Wear and Tees, and their fish stocks through the Industrial Revolution to the current day by using historical information. Before the 19th Century, Atlantic salmon and sea trout were widely distributed through all three catchments. The decline of fish stocks started by the early and middle 19th Century, and appeats to have been caused by multiple factors including coal and metal mining, industrial pollution, domestic pollution, construction of in-stream barriers, gravel mining, etc. From the 1960s, following decreased heavy industry, closure of mines and progressive improvements in sewage and wastewater treatment, salmon and sea trout started to recover in both Tyne and Wear catchment and stabilized in recent years. In the Tees catchment, salmon and sea trout numbers have slowly increased from the 1980s until today, rather than following the rapid trajectory of the Tyne and Wear. Knowledge of the decline and recovery of other fish species in the catchments is much more fragmentary. This study has revealed that the potential for recovery of anadromous salmonid stocks in post-industrial Pennine rivers with abundant salmonid spawning and nursery habitat, is driven by both accessibility and survival in the lower river, through barriers, pollution and predators (e.g. humans, seals). In addition, this study provides baseline water quality and fish community background information of multiple study sites including all those in both Chapters 4 and 5.

Chapter Three

Are national barrier inventories fit for stream connectivity restoration needs? A test of two catchments

This chapter contains a modified and extended version of the publication: Sun, J., Galib, S.M. & Lucas, M.C. (2020). Are national barrier inventories fit for stream connectivity restoration needs? A test of two catchments. *Water and Environment Journal*, **34**, 791–803. https://doi.org/10.1111/wej.12578

Statement

J. Sun and M. Lucas conceived the study and experimental design, J. Sun led the fieldwork with help from S. Galib between 2018 and 2019. J. Sun analysed the data and wrote the chapter, with comments provided by M. Lucas.

Summary

Catchment-scale river reconnection programmes require barrier inventories for restoration planning, yet barrier inventories are variable in extent and quality internationally. To test the degree to which barrier databases, in this case for England, are fit for purpose, a comparison was made of the national database (mostly originating from desk-study) for two catchments, the Wear and the Tees, against detailed walkover surveys. A total of 701 km (32.8%) of stream length were surveyed, stratified by stream order, altitude and subcatchment and recorded natural and artificial barriers. Only 22.7% of barriers identified in the walkover survey were present in the national database, including low-head (<5 m) artificial structures (32.3% representation), artificial barriers ≥5 m (14.3% representation) and culverts (0% representation). 18.9% of artificial barriers in the national database were found, during field survey, to have been breached naturally. Mean densities of artificial barriers were 0.68 barriers km^{-1} and 0.45 barriers km^{-1} in the Wear and Tees respectively, significantly higher than in the national database. Stream connectivity restoration in England may be hampered by the incomplete national barrier inventory. It is recommended that careful checks of barrier inventories are made as they are developed internationally and that these are regularly updated.

3.1 Introduction

Habitat fragmentation and loss of connectivity due to anthropogenic river barriers is one of the major impacts of humans on rivers (Chapter 1) and was one of the primary causes of the declines of migratory fish species in industrialized Northeast English rivers (Chapter 2). Artificial obstacles such as dams, weirs and sluices along rivers have been constructed to control floods, provide water for human consumption, irrigation and power supply (Jackson and Marmulla, 2001; Birnie-Gauvin *et al.*, 2017c; Galib *et al.*, 2018). Culverts and fords have been built to provide transport crossings or to route water through urban environments (Warren and Pardew, 1998; Price *et al.*, 2010). In-stream barriers, whether artificial or natural (e.g. waterfalls, glacial sediment plugs) can interrupt longitudinal and lateral connectivity, and so alter hydrology, sediment transport, nutrient flow and the movement of biota (Mueller *et al.*, 2011; Grill *et al.*, 2015). Natural barriers such as waterfalls can affect the biogeography, genetic structuring and diversity of organisms by limiting their dispersal, and partially or completely isolating populations, facilitating local adaptation (Whiteley *et al.*, 2010; Torrente-Vilara *et al.*, 2011). It is the density, distribution and nature of artificial obstacles that causes concern for damaging impacts to natural river processes and the ecosystems that are inherently linked to these (Lehner *et al.*, 2011; Jones *et al.*, 2019; Belletti *et al.*, 2020).

Removal or mitigation of anthropogenic barrier effects along rivers is a major aspect of river restoration programmes (Kemp and O'Hanley, 2010), including in Europe where large amounts of river infrastructure were installed during the Agricultural and Industrial Revolutions, some of which is now redundant (Birnie-Gauvin *et al.*, 2017c). Hydromorphology, comprising a stream section's hydrological regime, continuity and morphological condition, is an element of quality assessment under the Water Framework Directive (WFD) in European Union member states. In multiple EU states many rivers are failing, or at risk of failing, to reach good ecological condition due to impaired hydromorphological quality (Atkinson *et al.*, 2018; Jones *et al.*, 2019). River obstacles can alter habitats, disrupt dispersal between habitat patches, restrict or prevent migration and eventually lead to a decline in the abundance of sensitive species and biological diversity (Favaro *et al.*, 2014; Louca *et al.*, 2014; Birnie-Gauvin *et al.*, 2017a). Populations of diadromous fishes such as European eel (*Anguilla anguilla*) and Atlantic salmon (*Salmo salar*) have reduced significantly at least in part due to the impacts of artificial barriers (Parrish *et al.*, 1998; Piper *et al.*, 2013).

Globally, most large dams are recorded in databases (Lehner *et al.*, 2011; Grill *et al.*, 2015), and their impacts on river systems are well studied (Van Looy *et al.*, 2014). There are fewer such databases for small-scale barriers (but see Sheer and Steel, 2006; Januchowski-Hartley *et al.*, 2013; Atkinson *et al.*, 2018; Jones *et al.*, 2019; Belletti *et al.*, 2020) and they are mostly incomplete. Jones *et al*. (2019) found that the current barrier databases for Great Britain underestimated man-made barrier numbers by 68%, mostly due to under-recording of small barriers. Although small-scale barriers such as weirs, ramps and fords may have lesser impacts on biota per location than large dams, low-head barriers are much more abundant (Januchowski-Hartley *et al.*, 2013), and their cumulative effects on biota may be significant (Lucas *et al.*, 2009; Kemp and O'Hanley, 2010).

Globally there are 16.7 million reservoir impoundments, and 99.5% are small structures (reservoir surface area < 0.1 km^2) (Lehner *et al.*, 2011). According to a geographic information system (GIS) based desk study of maps (Entec, 2010), there are nearly 25,000 weirs and similar structures in rivers of England and Wales, of which 3000 of the barriers need connectivity restoration to meet EU WFD targets (Elbourne *et al.*, 2013). However, in order to mitigate the negative impacts of in-stream barriers, an effective strategy for river reconnection is needed as part of the restoration process (Kemp and O'Hanley, 2010; Tummers *et al.*, 2016). To do this barriers need to be mapped, measured, categorised and a barrier inventory generated (Januchowski-Hartley *et al.*, 2013; Atkinson *et al.*, 2018). The inventory can be used to prioritise which obstacles to remove or mitigate, depending on modelled benefits, restoration costs and objectives (King *et al.*, 2017). For river management, an inadequate restoration plan may lead to inefficiencies or waste of effort (Kemp and O'Hanley, 2010), and the accuracy of barrier inventories can directly affect connectivity restoration planning. So it is necessary to understand the true numbers, distribution and types of in-stream barriers of whole catchments for effective river connectivity restoration.

Across Europe there is much variability in the extent to which river barriers have been mapped and recorded (Garcia de Leaniz *et al.*, 2018). England is regarded as having one of the more complete and up-to-date barrier databases, originating from a desk-based study to map hydropower opportunities (Entec, 2010; Jones *et al.*, 2019). Ground-truth comparison of the Great Britain barrier database surveyed under 0.2% of stream length at 1:250,000 resolution, stratified across Great Britain (Jones *et al.*, 2019), with the possibility that more intensive validation surveys at the individual catchment level might generate different outcomes. To test the degree to which current national river barrier databases, in this case for England, may be fit for river-connectivity restoration purposes, intensive, stratified walkover surveys of two medium-sized catchments was carried out and compared them with the national river barrier database. Since one aim of this study was to measure stream connectivity for biota, especially fish, the occurrence and characteristics of in-river obstacles of natural and anthropogenic origin was recorded, as well as the existence and typology of fish passage devices and barrier removals.

3.2 Methods

3.2.1 Study area

The Rivers Wear and Tees were chosen for study because they are medium-sized catchments, somewhat typical of the variable topography and land uses occurring across large parts of Great Britain (Figure 3.1). They are also recovering post-industrial rivers, central to the thesis objectives of examining processes in the decline and recovery of native fish communities in post-industrial rivers of Northeast England (Chapter 1). The Wear and Tees are 110-km long and 160-km long respectively, both rising in the Pennine Hills and flowing eastwards to the North Sea. The lower reaches of both rivers pass through agricultural, industrial and urban areas, and the upper parts of the catchments were heavily exploited for metal mining in the 17th-19th centuries (Chapter 2). Coal mining and processing occurred widely through the lower and middle Wear catchment in the 18th-20th centuries. Water storage reservoirs occur in the catchments of both rivers (Wear: Burnhope Reservoir, Waskerley Reservoir and Tunstall Reservoir; Tees: Cow Green Reservoir, Selset Reservoir, Balderhead Reservoir, Blackton Reservoir, Hury Reservoir, Hurworth Burn Reservoir and Crookfoot Reservoir), especially the Tees, where they were built, in part, for maintaining industrial water supply to downstream reaches (see section 2.3.3.2). Large parts of the catchments are agricultural but they also have an extensive road and rail network, including river crossings, a proportion of which are disused transport routes originating during the Industrial Revolution. There is also a legacy of agricultural and industrial mills and weirs, almost all of which no longer serve their original purpose, but many are now linked to or near residential dwellings. This river infrastructure is similar in diversity and origins to much of that which developed in Britain and across Europe in the Agricultural and Industrial Revolutions (Downward and Skinner, 2005). Both rivers have recovering Atlantic salmon populations, following dramatic reductions in industrial and urban pollutant loadings in recent decades, although the Tees' recovery has been slow, probably due to a tidal barrage opened in 1995 (see sections 2.3.2.5 and 2.3.3.5).

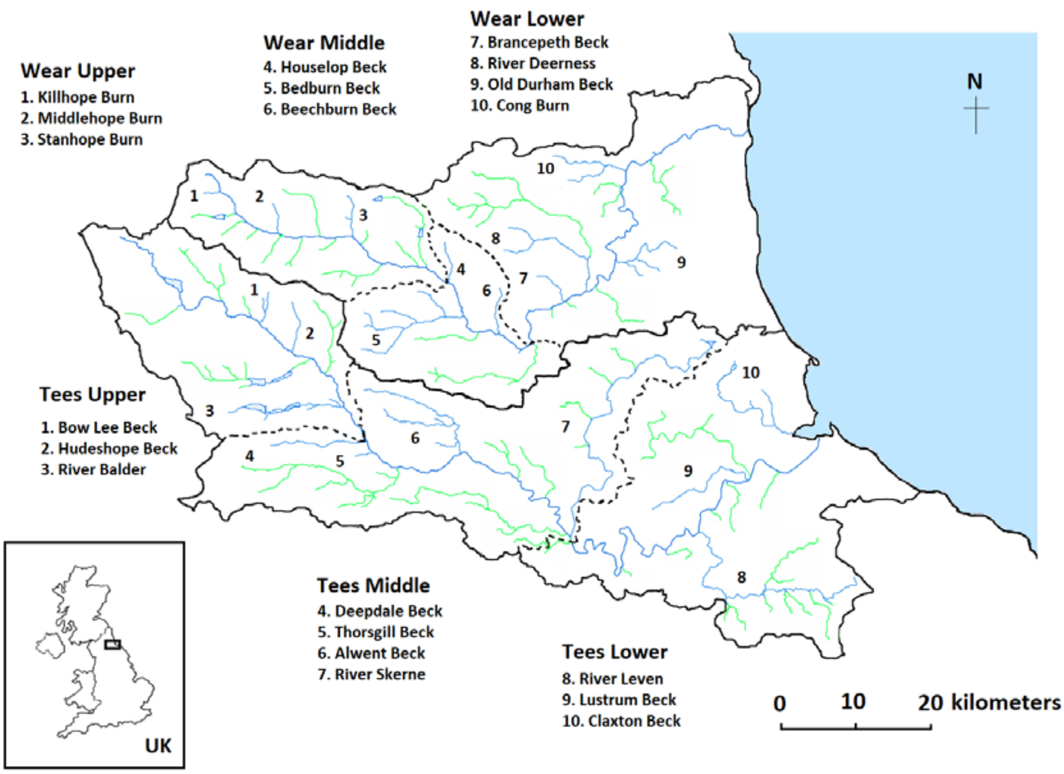


Figure 3.1 The location of the Wear and Tees catchments including their sub-catchments in England, as well as the location of field surveyed rivers (blue). The main River Wear and River Tees in each sub-catchment was also surveyed.

3.2.2 National river barrier database

In England, the national river barrier inventory used for management and longitudinal connectivity restoration planning was produced, and is held and managed, by the Environment Agency (EA) of England (Jones *et al.*, 2019). The EA barrier database was originally created from a desk-based study to map hydropower opportunities at river channel barriers across England and Wales (Entec, 2010), generally at sites having an in-channel drop greater than 1 m. The dataset of barrier locations was derived from an Ordnance Survey (OS) Master Map (Entec, 2010). Any structure on the map, passing across the river channel and listed as a dam, weir or waterfall was identified and mapped in the database. Therefore the database includes natural and anthropogenic barriers. Barrier height information was extracted from LiDAR (Light Detection and Ranging) and SAR (Synthetic Aperture Radar) datasets. Subsequently the EA has added sites to this database as they have been identified, particularly tidal water management sluices, and additional artificial barriers identified by local EA teams. The EA barrier inventory dataset used in this study was the same as that in Jones *et al.* (2019), generated in January 2018.

3.2.3 Independent barrier validation – stratified walkover surveys

In order to provide a quality assessment of the national barrier inventory, walkover surveys, stratified by stream order, altitude and position within the catchment (Jones *et al.*, 2019) were carried out in order to record natural and anthropogenic barriers. Only permanently-flowing streams were surveyed. Since the context of this study was from a longitudinal connectivity restoration viewpoint, particularly as regards fish passage, obstacles that had the potential to limit upstream movement of fish at normal to low flows (~Q_{50}-Q_{90}) were recorded, while acknowledging that maintaining free downstream-migration passage is also important (Silva *et al.*, 2018). Obstacles to free movement of fishes depend on obstacle characteristics (especially height and gradient), fish species and environmental conditions (Kemp and O'Hanley, 2010; Barry *et al.*, 2018). During the survey, any artificial structure having a vertical or steeply-sloping (> 45 degrees) step, exceeding 0.2 m in height, was regarded as a potential obstacle to weakly-swimming taxa (Utzinger *et al.*, 2008; Tummers *et al.*, 2016). More gently sloping structures (e.g. culverts) without an obvious step were regarded as potential obstacles if they had a fall in height along their length exceeding 0.5 m and/or were very constrained (e.g. pipe culverts), and/or very shallow (< 3 cm at ~Q_{90}, e.g. many artificially-lined culverts; Tummers *et al*., 2016). This is a simpler framework than the SNIFFER and ICE rapid barrier assessment methods (Barry *et al.*, 2018) but deliberately so as even small obstacles may impact dispersal and recolonization of non-jumping fish species (Tummers *et al.*, 2016). Any natural waterfall or cascade exceeding 0.5 m high was regarded as a potential obstacle, as well as extensive bedrock sills with water depth < 3 cm. Although passability of natural barriers will vary between fish taxa, it was felt that this enabled a reasonable compromise to be applied practicably for this study. River restoration projects rarely seek to alter natural connectivity barriers, such as waterfalls, and so barrier inventories tend only to record obstacles of anthropogenic origin. This study recorded natural obstacles in order to provide a context to the distribution of anthropogenic barriers, and to enable comparison to the national inventory of such barriers. Further, understanding the distribution of both natural and anthropogenic barriers in a catchment can play a role in better catchment planning for restoration of migratory species populations (Silva *et al.*, 2018) and/or for limiting the spread of invasive species by managed habitat fragmentation (Rahel and McLaughlin, 2018).

Walkover surveys of almost all but the smallest catchments rely upon subsampling (Jones *et al.*, 2019), or progressive development of a database over a period of many years (Sheer and Steel, 2006). In this study the OS Open Rivers (1: 25,000) GIS was used for river mapping and subsampling the Wear and Tees for walkover surveys. On this system and scale, first-order streams (Strahler, 1957) normally had a field-observed wetted channel width of less than 3 m (J. Sun, pers. obs.). Typically, stream reaches in the lower resolution (1: 250,000) European Catchments and Rivers Network System (ECRINS: European Environment Agency, 2012) database are recorded as a Strahler stream order lower than in this study, reflecting the lower spatial resolution of the ECRINS database. Thus, most first order streams recorded in this study do not exist in ECRINS, and first order streams listed for the Wear and Tees in Jones *et al.* (2019) which employed ECRINS, were typically recorded as second order streams in this, finer resolution, study.

In order to stratify walkover surveys across a range of stream orders, altitudes and sections within the Wear and Tees catchments, each of these watersheds was split into upper, middle and lower subcatchments (Figure 3.1) based upon EA operational catchment areas. Three or four tributaries were quasi-randomly selected from each operational catchment for conducting the walkover survey. Each of these provided multiple sections of Strahler first- to fourth-order streams to survey. Besides these tributaries, the main channels of the Rivers Wear, Tees, and sections of the Browney (Wear), Skerne (Tees) and Leven (Tees) were included in the walkover survey, in order to sample extensive lengths of stream orders 4 and 5. This is because longitudinal connectivity obstacles on main river channels are particularly important to identify, especially for diadromous migratory fish (Silva *et al.*, 2018), even if they tend to be well recorded in existing barrier inventories (Jones *et al.*, 2019). Although the Browney (containing River Deerness), Skerne and Leven were defined as operational catchments by the EA, the Browney was categorized in the Lower Wear, the Skerne in the Middle Tees Catchment and the Leven in the Lower Tees subcatchments based on their geographic locations (Figure 3.1).

Additionally any online, large artificial water bodies (> 10 ha) evident on 1:25,000 maps, and with an obvious dam, were visited and obstacle characteristics recorded by visual inspection, reference to maps and any information available on their construction.
Field surveys were carried out by an experienced team. For each tributary selected, the survey normally covered the whole stream length (and for all adjoining streams) from the main river confluence upstream towards the source, to the limit of the channel evident on OS Open Rivers 1: 25,000. The location (British national grid reference) and altitude (m above sea level) of physical obstacles, both natural and artificial, were recorded as they were encountered. The barrier type, height, gradient, pool depth (immediately below obstacle) and length (for culverts and concrete channels) were measured and a brief description made. Photographs for each barrier, with a scale bar alongside, were taken.

At any artificial obstacles where modification had occurred with the apparent aim of improving river connectivity for fishes (fishways and other passage easements) information on that was gathered from field measurements, as well as from EA and Rivers Trust records. Sites where barriers had existed in the recent past (national database) but had collapsed, breached or been removed deliberately within the areas surveyed was also recorded. Appendix 1 within this thesis describes the development of, and provides an electronic link to a Master database of up to date barrier and fishway locations, characteristics and photographs in the Wear and Tees catchments, resulting from integration of all available information, for the benefit of catchment restoration management by EA and key stakeholders such as Rivers Trusts.

3.2.4 Data analysis

Barrier data from the field were entered into a spreadsheet inventory. Each barrier was given a unique code and associated with a barrier photograph. The Strahler stream orders of all channel segments in the two catchments were identified using OS Open Rivers (1:25,000). The cumulative distances field surveyed and the proportion of field-surveyed river length in each stream order were calculated by QGIS (version 2.18.4) using river segment lengths from OS Open Rivers.

Barriers from the EA national database identified as occurring in non-qualifying habitat (not on OS 1: 25,000 Open Rivers network or found to be dry, so not representing permanent aquatic habitat) were excluded from analysis. Artificial barrier density was calculated for each river section for a given stream order, using the total number of artificial barriers divided by total river length (km) in that section.

Artificial and natural barrier densities in the national database was compared with field surveyed barrier densities for the same river sections. Artificial barrier heights measured in the field survey were compared across the two catchments and also with the distribution of barrier heights from the national database. Where data were not normally distributed they were transformed log (x+1) before statistical comparison. ANOVA was used to compare barrier densities between stream orders, and between upper, middle and lower catchment areas. *t*-tests were used to compare mean barrier height between the catchments. Paired *t*-tests were used to compare barrier heights and densities between the walkover survey data and national database. All tests were run in SPSS (Version 22).

The overall barrier abundance of the whole catchment was estimated by two methods. In Method one (simple uprating), barrier density was calculated for each stream section having a particular Strahler stream order, then mean barrier density across all surveyed stream sections (Wear n = 83, total length 280 km; Tees n = 62, total length 421 km) was multiplied by the total stream length in the catchment. In Method two (uprating by stream order proportions) the same calculation was applied to estimate total numbers of barriers for total length of each Strahler stream order in a catchment and these subtotals for Strahler stream orders were summed to generate a value for the entire catchment.

3.3 Results

3.3.1 River Wear catchment

In the Wear, 752 km (to nearest km) of stream channel length were mapped from OS Open Rivers 1: 25,000 (1st order, 330 km; 2nd order 202 km, 3rd order, 75 km, 4th order 44 km, 5th order 100 km) and a total of 280 km (37.3%) of the Wear catchment stream length was field surveyed. Across field-surveyed reaches of the Wear, 364 barriers were recorded, 41.2% (n = 150) of which were artificial barriers (Figure 3.2) and 58.8% (n = 214) were natural barriers (waterfalls and cascades) (Figure 3.2, 3.3). Mean artificial barrier height was 1.40 m (95% CI Bootstrap: 0.64 - 2.38 m), and mean natural barrier height was 1.31 m (95% CI Bootstrap: 1.02 - 1.58 m). Most barriers were located in first and second order streams, comprising 78% (n = 117) of artificial barriers and 79% (n = 169) of natural barriers. Artificial barriers were most frequent at low altitudes, while the opposite occurred for natural barriers (Figure 3.3). Among artificial barriers within the field survey area, 19.2% (n = 29) had a fishway or other passage mitigation, seven further barriers had been deliberately removed for connectivity restoration and another 11 washed away (Figure 3.4).

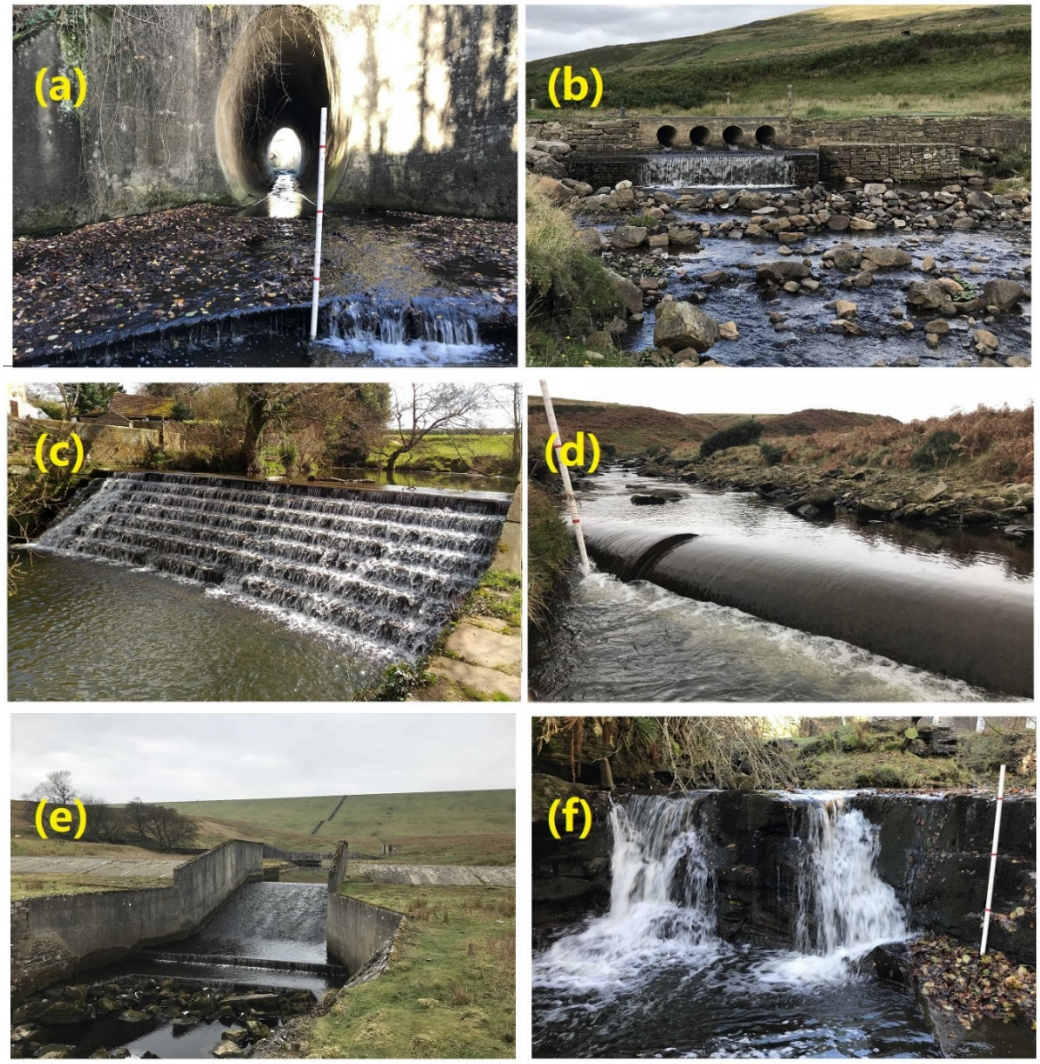


Figure 3.2 Different types of river barrier identified during the walkover survey. (a) culvert in Houselop Beck; (b) pipe culvert and concrete weir in Hudeshope Beck; (c) stepped weir in the River Leven; (d) metal pipe in Stanhope Burn (e) Balderhead Reservoir dam and its spillway in the River Balder (f) waterfall in Middlehope Burn. Red markers on the measuring pole are 0.5-m intervals.

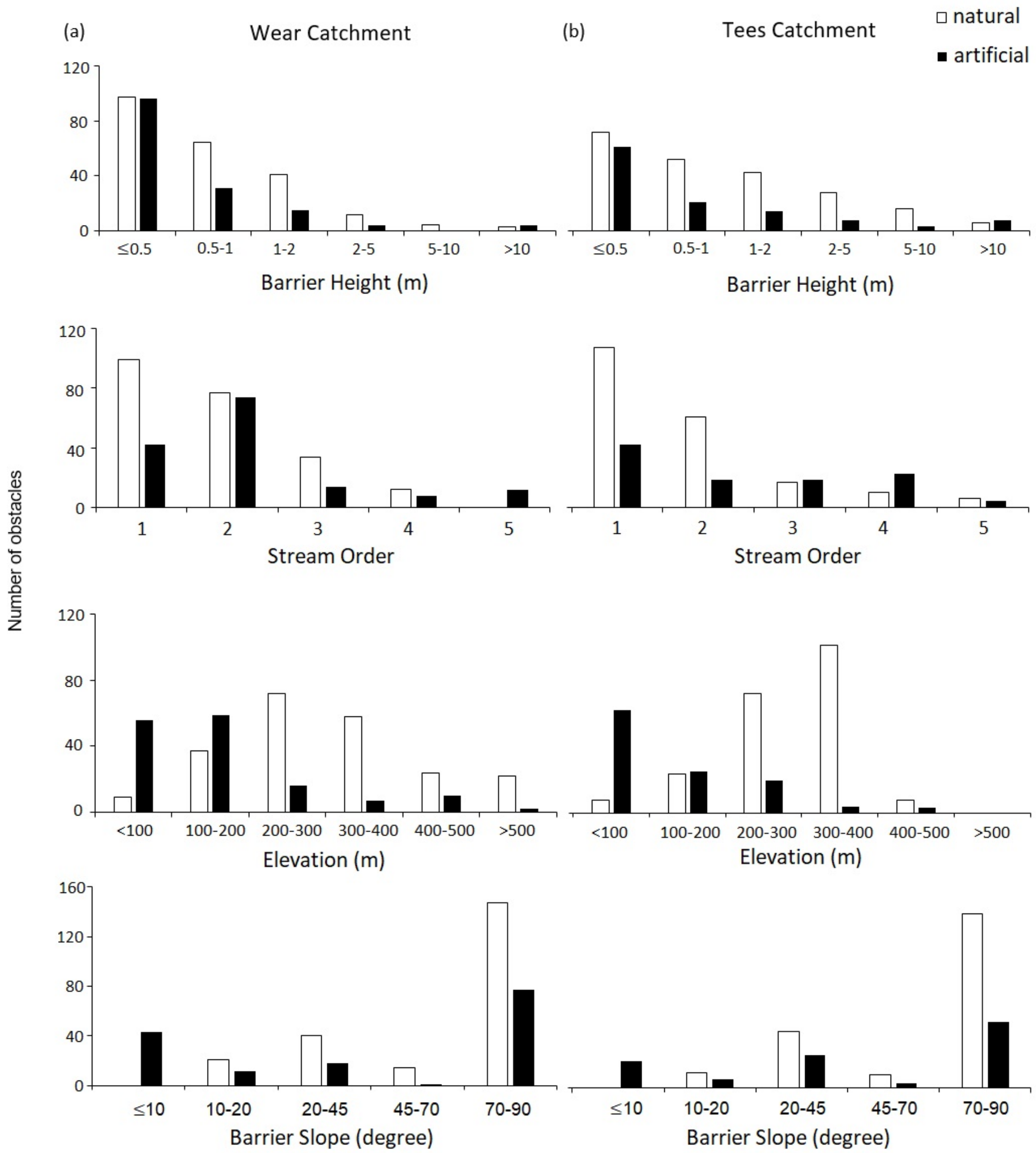


Figure 3.3 Natural and artificial barrier height, stream order, barrier elevation and slope on (a) the Wear and (b) the Tees catchment.

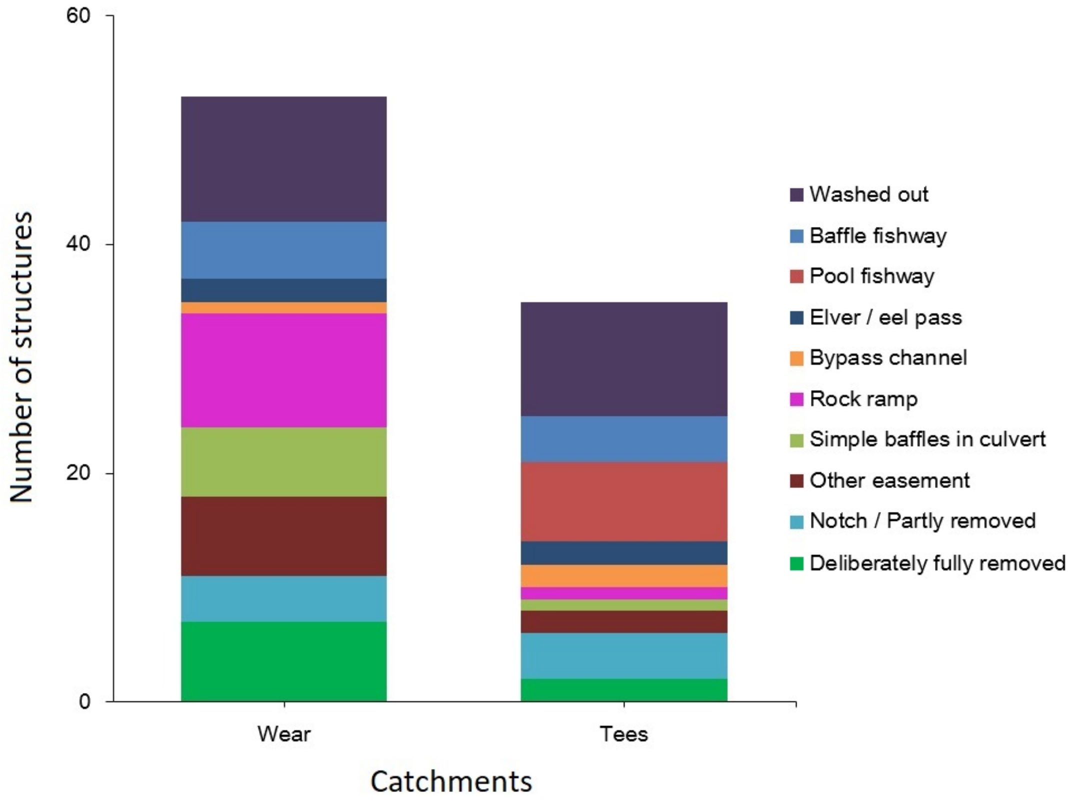


Figure 3.4 Numbers of artificial barriers deliberately removed for connectivity restoration, washed out, or fitted with fish passess in the Wear and Tees. Elver / eel pass refers to bristle and /or studded substrate. ‘Other easement’ refers mainly to pre-impoundments built downstream of the main obstacle to raise the water levels and facilitate passage by jumping species.

The mean artificial barrier density of the Wear catchment was 0.68 barriers/km (95% CI Bootstrap: 0.47 - 0.91 barriers/km). Barrier density did not differ across stream orders 1-3 (ANOVA, $F_{2,74}$= 2.600, p = 0.081), for which sufficient samples sizes were available for analysis. Lower barrier densities occurred at stream orders 4 and 5 (Table 3.1, not statistically tested due to small sample size). The density of artificial barriers did not differ between the upper, middle and lower Wear subcatchments (ANOVA, $F_{2,80}$ = 1.657, p = 0.197). The total number of artificial barriers in the Wear, estimated by simple uprating, using an average artificial barrier density of 0.68 across the entire field surveyed area was 512 (Table 3.2). The total number of artificial barriers estimated by Method 2, summing the estimated numbers for all Strahler stream orders was 479 (Table 3.2).

Table 3.1 Summary of fieldwork surveyed river length (km) under each stream order in the Wear catchment, and the mean artificial barrier density at each stream order.

Catchment	Stream order	River length (km)	River section (*n*)	Artificial barrier number (*n*)	Artificial barrier density per section (n/km)	
					Mean	SD
Wear upper	1	14.5	22	4	0.24	0.64
	2	12.3	7	14	1.54	0.98
	3	8.5	2	2	0.15	NA
	4	10.5	1	5	0.47	NA
	5	17.9	1	3	0.17	NA
Wear middle	1	10.2	13	14	1.04	1.54
	2	20.9	7	19	0.37	0.66
	3	9.4	2	1	0.10	NA
	4	8.1	1	2	0.25	NA
	5	16.9	1	4	0.24	NA
Wear lower	1	28.7	15	24	0.80	1.04
	2	42.9	7	40	1.19	0.74
	3	7.8	2	10	1.18	NA
	4	6.2	1	1	0.16	NA
	5	65.3	1	7	0.11	NA
Wear overall	1	53.5	50	42	0.62	1.11
	2	76.1	21	73	1.03	0.94
	3	25.7	6	13	0.48	0.72
	4	24.9	3	8	0.29	0.13
	5	100	3	14	0.17	0.05
Combined		280.2	83	150	0.68	1.03

Table 3.2 Estimated numbers of artificial barrier numbers in the Wear and Tees using Method 1 (average density across all stream segments in field survey zone multiplied by total catchment stream length) and Method 2 (sum of estimated barrier numbers for combined length of each Strahler stream order). Totals are shown in bold.

Catch-ment	Method	Stream order	Length	Density	95% CI		Estimated number	95% CI	
Wear	1	total	752.323	0.68	0.47	0.91	**512**	354	685
	2	1	330.602	0.62	0.33	0.96	205	109	317
		2	202.32	1.02	0.63	1.44	206	127	291
		3	74.898	0.44	0.08	1.02	36	6	84
		4	44.418	0.29	0.16	0.47	13	10	18
		5	100.085	0.13	0.1	0.16	17	15	19
		combined					**479**	267	729
Tees	1	total	1388.727	0.45	0.29	0.62	**625**	403	861
	2	1	667.429	0.58	0.3	0.89	387	200	594
		2	321.13	0.23	0.1	0.43	74	32	138
		3	182.513	0.46	0.15	0.87	84	27	159
		4	97.136	0.28	0.05	0.51	27	5	50
		5	120.519	0.03	0	0.05	4	0	6
		combined					**576**	264	947

The EA's national barrier database contained 254 barriers for the Wear, 69 (artificial and natural) of which were within this study's field-surveyed areas (Figure 3.5). The national database included one of four barriers larger than 10 m in height (Figure 3.6), none of which incorporated fishways. Since 15 of the artificial barriers in the national database for the Wear had been washed away or removed already, only 54 barriers (33 artificial and 21 natural barriers) were valid in the national database for the field-surveyed area (Figure 3.6). The artificial barrier density calculated from the national database (0.04 barriers/km) was significantly lower compared with the walkover-surveyed barrier density (paired *t*-test on transformed data, t_{82} = 6.630, p < 0.001). Overall, 78.0% (n = 117) of artificial barriers and 90.2% (n = 193) of natural barriers were missed in the national database for walkover-surveyed areas of the Wear (Figure 3.3). Artificial barriers in the national database for the Wear were exclusively weirs, but approximately equal numbers of weirs, culverts and bridge aprons occurred in the walkover survey (Figure 3.5). None of the small cascades and waterfalls (< 2 m high, n = 192) identified in field walkovers were recorded in the national database. A significant difference occurred between walkover survey barrier (natural and artificial combined) heights (mean ± SD, 1.33 ± 3.79 m) and national database barrier heights (4.10 ± 3.89 m) (independent t-test on transformed data, t_{422} = 9.237, p < 0.001), showing the national dataset concentrates on larger obstacles.

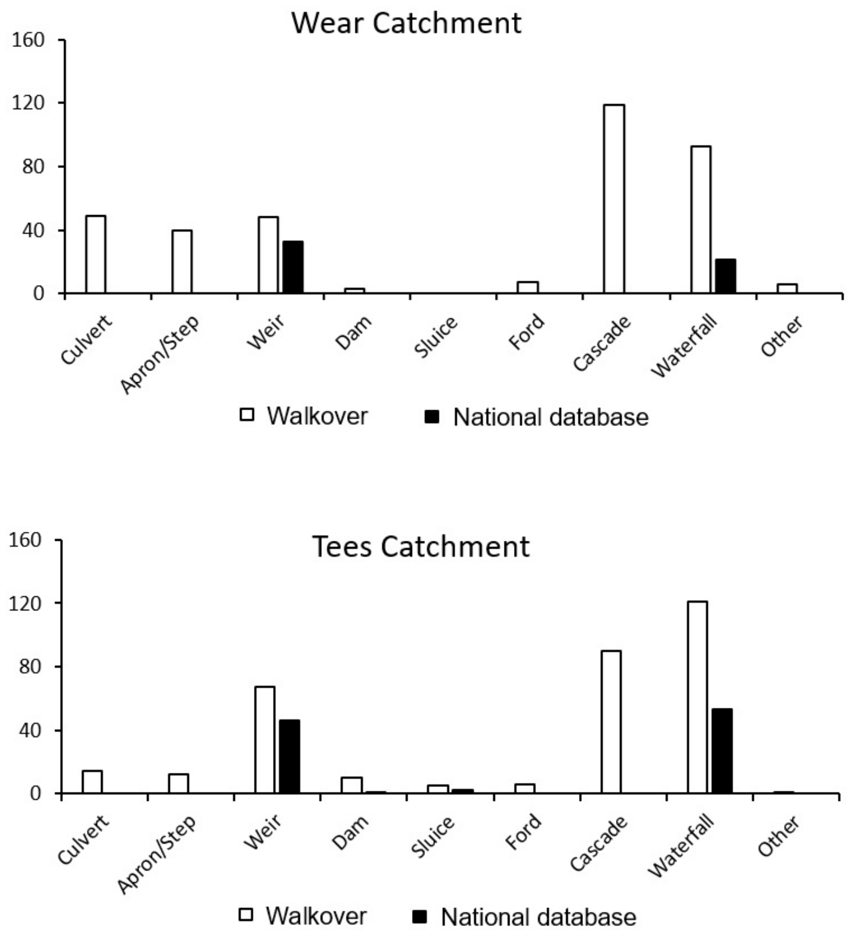


Figure 3.5 Different barrier types recorded in the walkover survey database and EA database on (a) the Wear and (b) the Tees catchment. “Other” refers to: collapsed bridge (n = 1), spillway (n = 4), concrete channel (n = 1) and tidal barrage (n = 1).

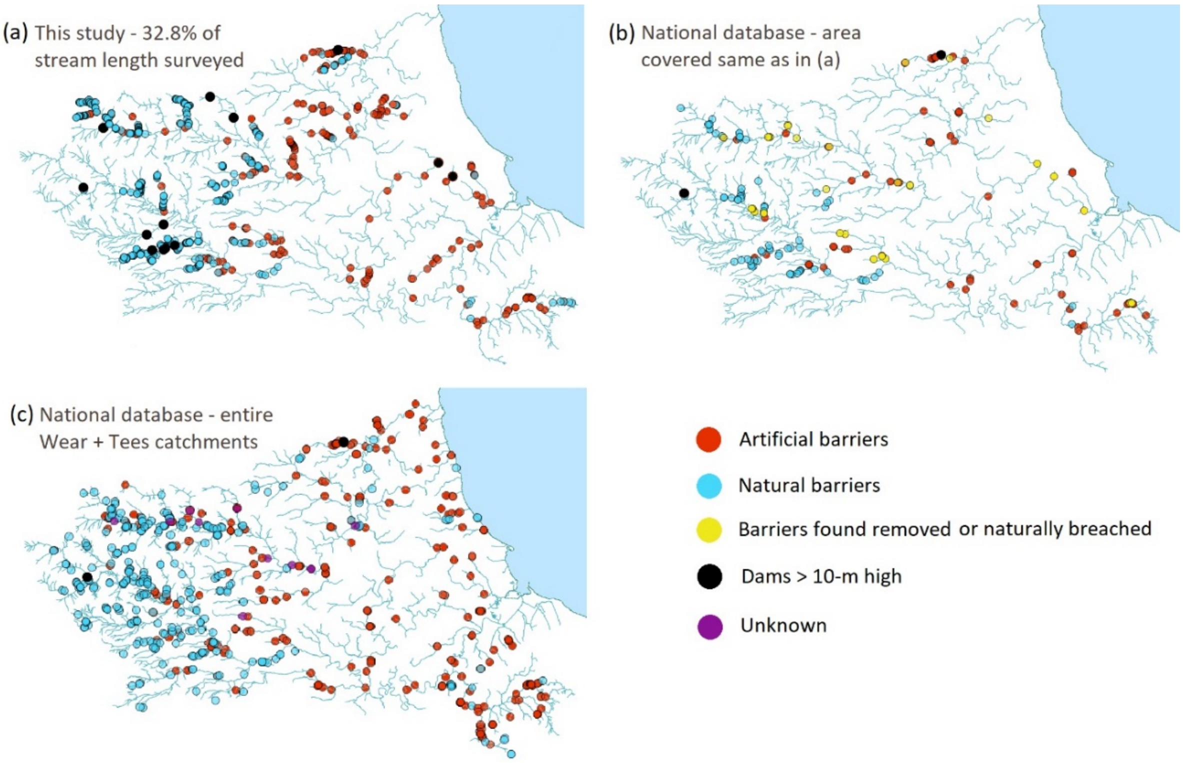


Figure 3.6 Locations of different types of barrier recorded in (a) walkover survey database, (b) National database under same walkover survey range and (c) National database for the entire Wear and Tees catchments. Purple circles: barriers classified as unknown in the national database.

3.3.2 River Tees catchment

In the Tees, 1389 km of stream channel length were recorded in 1: 25,000 OS Open Maps (1st order, 667 km; 2nd order 321 km, 3rd order, 183 km, 4th order 97 km, 5th order 120 km) were recorded. A total of 421 km river length were walkover-surveyed, covering 30.3%of stream length in the whole Tees catchment. Across the field-surveyed area, 322 barriers were recorded, of which 65.1% (n = 211) were natural and 34.9% (n = 111) were artificial barriers (Figure 3.3). Artificial barriers were most frequent at low altitudes, while the opposite occurred for natural barriers (Figure 3.3). Mean artificial barrier height was 2.95 m (95% CI Bootstrap: 1.73 - 4.45 m), and mean natural barrier height was 2.28 m (95% CI Bootstrap: 1.78 – 2.96 m). Heights of natural (Independent t-test on transformed data, t_{435} = 4.109, p < 0.001) and artificial barriers (Independent t-test on transformed data, t_{260} = 2.848, p < 0.001) were significantly higher in the Tees than Wear catchment. Most (82.9%)

natural barriers in the Tees were located in first and second order streams. In field-surveyed reaches of the Tees, 67.6% (n = 75) of artificial obstacles were weirs and dams. Overall, 16.2% (n = 18) of artificial barriers surveyed had a fishway or other passage mitigation (Figure 3.4). Two further barriers had been deliberately removed for connectivity restoration and another 10 had collapsed (Figure 3.4).

The mean artificial barrier density of the Tees catchment was 0.45 barriers/km (95% CI Bootstrap: 0.29 - 0.62 barriers/km). Barrier density did not differ across stream orders 1-3 (ANOVA, $F_{2,53}$ = 0.745, p = 0.479). High order streams tended to have lower densities of barriers (Table 3.3). There was no difference in the density of artificial barriers between the upper, middle and lower Tees subcatchments (ANOVA, $F_{2,59}$ = 8.38, p = 0.410). Using the global average artificial barrier density of 0.45 barriers km^{-1} uprated by total stream length, the total number of artificial barriers in the Tees was estimated as 625 (Table 3.2), while summation of the subtotals per Strahler stream order gave an estimated total of 576 (Table 3.2).

In the national database, a total of 113 barriers were recorded within this study's field survey area of the Tees. The national database did not record eight dams higher than 10 m (none of which have fishways) that exist within the Tees catchment. As 11 of the artificial barriers in the national database had been removed for river restoration purposes or washed away (Figure 3.4), 102 barriers (49 artificial and 53 natural barriers) were valid in the national database (Figure 3.6). The artificial barrier density in the Tees catchment from the national database (0.09 barriers km^{-1}) was significantly lower than for the same stream segments in the walkover survey (paired t-test on transformed data, t_{61} = 5.317, p < 0.001). 55.9% (62) of artificial barriers and 74.9% (158) of natural barriers were missed in the EA database compared with the walkover survey (Figure 3.6). None of the culverts (n = 14) or aprons (n = 9) identified in the field survey were recorded in the national database. Mean barrier height (4.80 ± 4.49 m) from the national database was significantly higher compared to the walkover survey database (2.49 ± 6.05 m) within the same surveyed areas (independent t-test on transformed data, t_{429} = 7.482, p = 0.01).

Table 3.3 Summary of fieldwork surveyed river length (km) under each stream order in the Tees catchment, and the mean artificial barrier density at each stream order.

Catchment	Stream order	River length (km)	River section (*n*)	Artificial barrier number (*n*)	Artificial barrier density per section (n/km)	
					Mean	SD
Tees upper	1	15.0	17	3	0.50	1.10
	2	23.6	7	4	0.27	0.52
	3	23.4	2	8	0.32	NA
	4	20.5	1	1	0.05	NA
	5	14.0	1	0	0	NA
Tees middle	1	41.6	9	32	0.86	0.78
	2	22.7	5	5	0.19	0.11
	3	49.0	2	11	0.37	NA
	4	0.0	0	NA	NA	NA
	5	37.5	1	2	0.05	NA
Tees lower	1	22.7	9	10	0.47	0.69
	2	32.7	4	9	0.23	0.36
	3	6.2	2	1	0.69	NA
	4	42.9	1	22	0.51	NA
	5	69.0	1	3	0.04	NA
Tees overall	1	79.3	35	45	0.58	0.94
	2	79.0	16	18	0.23	0.36
	3	78.6	6	20	0.46	0.44
	4	63.4	2	23	0.28	NA
	5	120.5	3	5	0.03	0.02
Combined		420.8	62	111	0.45	0.77

3.4 Discussion

This study provides a test of the adequacy of the English national barrier database for two typical medium-sized catchments, albeit neighbouring catchments within the same geographic region. Large-scale under recording of obstacles, including most large water storage dams was found. The study has generated the first intensive but, as yet still incomplete, inventory of artificial and natural barriers in the Wear and Tees catchments and provides a valuable resource for river restoration work in the future. Although artificial barrier density was greater at lower altitudes, generally in the middle and lower catchments, barriers were widespread throughout the catchments, reflecting the diversity of their origins. This study indicates that 77.3% of the in-stream barriers in both catchments were absent in the national database, including 68.6% of artificial barriers and 82.6% of natural barriers. The field-validated barrier densities are significantly higher by comparison with the EA national database barrier densities. The EA barrier inventory is likely to be one of the more complete inventories in Europe (Belletti *et al*., 2020; http://www.amber.international). So it also seems likely that in other countries where barrier inventories have been mapped by desk study there may be similar levels of error. It is estimated that 76%–98% of in-stream barriers in the Balkans were missing from the existing barrier inventory, also in Estonia (91%), Greece (97%), and especially in Sweden where 100% low-head structures were absent in the existing database (Belletti *et al.*, 2020).

A total of 13 artificial barriers taller than 10 m (nine in the Tees, four in the Wear) were recorded during the survey, but only two of these were in the EA national barrier inventory, even though almost all are water supply reservoirs, none of which have fish passage facilities. Three of these dams were present in the Global Reservoir and Dam (GRanD) database (Grill *et al.*, 2015) and hence in the database generated by Jones *et al*. (2019), which also contains one additional non-duplicated barrier from the EA national database. In the UK, the Inventory of Reservoirs Database contains 273 individual reservoirs, which account for 90% of UK reservoir storage (Durant and Counsell, 2018) but evidently, within the Wear and Tees catchments, most of these are not integrated into the EA's national barrier database. The UK's Inventory of Reservoirs Database was missing four dams with a height greater than 10 m compared to our database for the Wear and Tees. Thus, not only does the EA national obstacle inventory contain a small fraction of all artificial barriers, it also excludes some of the largest and most significant river barriers. Most of these large dams in the Tees and Wear are located in headwater valleys, where the majority of natural barriers also occur.

None of the large Tees/Wear dams have fishways. Although several fishways were incorporated into their dam designs when built over a century ago, they are now defunct (J.Sun, pers. obs.). It could be argued that fishways would be of little use at these headwater dams due to elimination, by the dams, of fluvial nursery habitat necessary for migratory salmonids (Silva *et al.*, 2018). These dams have also led to starvation of gravel transport to the river reaches immediately downstream, impacting habitat quality for salmonid spawning and other native rhithral biota (B. Lamb, pers. comm.). Efforts to reintroduce gravel downstream of Hury Dam on the River Balder, in recent decades, to enhance the Balder's ecology and salmonid spawning potential, have not appreciably increased juvenile salmonid densities there (Environment Agency, 2010, EA, unpublished data; Appendix I, Figure S3.1). On the Tees, the largest of these impoundments, Cow Green Reservoir, is also upstream of several large natural barriers that are impassable in an upstream direction by fish (Crisp, 1977; Crisp *et al.*, 1983; Armitage, 2006). Nevertheless, national barrier inventories must include all large obstacles, and most smaller ones, in order to be fit for purpose for river-basin planning activities. Although local EA officers are generally aware of these large dams, their absence on the national database probably reduces national and regional strategic consideration as to how to mitigate their ecological and environmental impact on the associated river systems. For example, historically the River Lune was once one of the River Tees' most important salmon spawning tributaries but now only a few percent of the Lune is accessible due to water storage dams (P. Frear, pers. comm.).

Fishways and other passage easements are the most common engineering mitigation for loss of river connectivity (Silva *et al.*, 2018). However, in order to restore river processes in fragmented rivers, removal of redundant barriers is increasingly used and recommended (Bednarek, 2001; Poff and Hart, 2002; Tummers *et al.*, 2016) because hydromorphic as well as ecological processes are reinstituted (Roni *et al.*, 2008; Birnie-Gauvin *et al.*, 2017a). In the field survey area only 21.5% (56/261, Wear and Tees combined) of artificial barriers had been mitigated with fishways / easements or removed. Only nine of the 261 structures (3.5%) in the survey areas across the two catchments had been deliberately removed. However, 21 weirs recorded on the EA's desk-study generated national database and within this study's walkover area were recorded as washed out by floods, or perhaps by other informal mechanisms (e.g. non-reported dismantling by humans). This represents 8.1% (21/261) of all artificial structures recorded. Many of these structures were old mill weirs, some centuries old and often of blockstone design, the remains of which were evident. The high energy of upland rivers such as the Wear and Tees during spate can breach such structures when not kept in good repair. Evidently a significant proportion of the artificial barriers listed in the English national barrier database are unlikely to be barriers any more, particularly within upland high-energy river systems. Atkinson *et al.* (2018) showed that river barrier inventories generated from mapping methods, as is mainly the case for the English river barrier inventory, must be validated by visiting all potential barriers identified by desk study.

Maintaining accurate and up-to-date river barrier inventories must be a priority for river reconnection restoration, for example to optimize the efficacy of barrier mitigation/removal actions at the catchment scale (King *et al.*, 2017; Barry *et al.*, 2018). Most ongoing stream reconnection actions in English catchments, including the Tees and Wear, are currently planned by regard to the potential for converting 'failing' WFD stream segments to 'good ecological condition' (see section 1.3) without fully considering the basin-wide distribution and characteristics of natural and artificial barriers. Because many river barriers in England are privately, rather than state-owned, and ownership is, in many cases, unknown or contested, barrier mitigations or removals frequently occur at sites where there is greatest facilitation by stakeholders and owners, not necessarily at the highest priority sites in restoration terms. This is certainly the case in the Wear and Tees, where much of the river restoration is carried out by 'third sector' rivers trusts (Wear Rivers Trust, Tees Rivers Trust – see section 1.5.1), often part-financed by state aid but where access and opportunity are major constraints in what can be achieved, and where (M. Lucas, pers. comm.).

In Great Britain, a recent study indicated that 68% of artificial barriers recorded in the field are missing from the existing database and a large proportion of the missing barriers are structures less than 1-m high (Jones *et al.*, 2019). That study adopted the coarser 1: 250,000 scale ECRINS GIS (European Environment Agency, 2012) for determining field surveys and missed most of the smaller stream channels which recorded as Strahler first order at 1: 25,000 mesh in this study. At 1: 250,000 Jones *et al*. (2019) validated 0.2% of river network, whereas at 1: 25,000 this study validated 37% and 30% by stream length of the Wear and Tees catchments respectively. The percentages of artificial barriers estimated to have been missed in the national barrier inventory for the Wear and Tees were 78% and 55.9% respectively. Despite the difference in spatial resolution and intensity of survey between these studies, under-reporting of artificial barriers for the Wear and Tees are not greatly different to the overall 68% under-reporting value estimated by Jones *et al*. (2019) for the whole of Great Britain and gives confidence in the validity of that estimate. The importance of spatial resolution for barrier inventories is highlighted by the fact that in this study over 70% of river networks for the Wear and Tees comprised first and second order streams, while for Ireland the value is 77% (McGarrigle, 2014). In an audit of the accessibility of juvenile Atlantic salmon habitat in the River Nore, Ireland, Gargan *et al.*, (2011) excluded first order streams and those with a gradient exceeding 4%, on the basis that those streams are used little by salmon. By contrast, first and second order coastal streams are widely used by sea trout *Salmo trutta* for spawning and nursery areas in Denmark (Aarestrup *et al.*, 2003). In the Wear and Tees catchments, first and second order streams provide important spawning and/or rearing habitat for fish species such as bullhead, stone loach, brown trout, and eel (Chapter 2). Clearly, the spatial resolution for barrier audits needs to take careful consideration of the environmental restoration objectives.

Although desk-study generation of barrier inventories using historic maps, overhead imagery and transport infrastructure routes is a useful tool (Januchowski-Hartley *et al.*, 2013; Atkinson *et al.*, 2018), there is a growing consensus that these must be validated by field-surveying (Atkinson *et al.*, 2018; Jones *et al.*, 2019). The easiest way of removing false-positives is to visit potential obstacles identified but this does not avoid missing artificial barriers not apparent from maps and overhead imagery, especially in urban or heavily tree-lined areas (Atkinson *et al.*, 2018). Despite catchment-scale walkover survey methods being time consuming, the method provides high-quality data to generate a reliable barrier for catchment-scale connectivity restoration. It was recommend that walkover surveys are undertaken, subcatchment by subcatchment, to develop comprehensive barrier inventories, which are regularly updated as barriers are added, removed or mitigated in order out to enable effective river-connectivity restoration planning and actions. Even when catchment barrier inventories are complete, periodic walkover audits, possibly supplemented by drones or other technology where topography allows, will need to be undertaken in order to take account of natural breaches and intentional removal of redundant obstacles.

3.5 Conclusions

This study provides evidence that the current national barrier inventory is highly incomplete by field validating two medium-sized catchments in NE England. This study showed that many artificial and natural barriers were missing from the national barrier inventory, resulting in lower barrier densities in the EA national database compared with field-validated barrier densities. Partial or complete failure in restoring stream connectivity may be expected if an incomplete barrier inventory is used. In addition, this study showed that only small amounts of artificial barriers had been removed or mitigated (by addition of a fish pass or easement) in both catchments. In order to achieve good ecological status (*sensu* WFD), intensive connectivity restoration is needed. Furthermore, this study showed that a large proportion of the missing barriers were located in first and second order streams (at 1: 25,000 mesh). Since these streams provide important spawning and rearing habitat for a number of fish species, they should not be missed out from river barrier surveys.

Chapter Four

Rapid response of fish and aquatic habitat to removal of a tidal-limit barrier

This chapter contains a modified and extended version of the publication: Sun, J., Galib, S.M. & Lucas, M.C. (2021). Rapid response of fish and aquatic habitat to removal of a tidal barrier. *Aquatic Conservation: Marine and Freshwater Ecosystems*.

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Statement

J. Sun and M. Lucas conceived the study and experimental design, J. Sun led the fieldwork with help mainly from S. Galib between 2017 and 2019. A wider range of 'minor' fieldwork support providers are listed under 'Acknowledgements' at the start of the thesis. J. Sun analysed the data and wrote the chapter, with comments provided by M. Lucas and S. Galib.

Summary

River barrier removal is used increasingly as a conservation tool to restore lotic habitat and river connectivity, but evidence of its efficacy is incomplete. This study used a before-after methodology to determine the effects of removing a tidal-limit barrier on the fishes, macroinvertebrates and habitats of an English coastal stream. Following barrier removal, habitat diversity increased immediately upstream and remained similar downstream. Mobilised silt altered the substrate composition immediately downstream, but this was temporary, as silt was flushed out the following winter. Changes to macroinvertebrate communities occurred upstream and downstream of the former barrier but these were transient. A dramatic and sustained increase in fish density occurred immediately upstream of the barrier after its removal, but effects downstream were minor.

The fish community upstream changed, largely due to rapid recolonization by endangered European eel (*Anguilla anguilla*), especially of elver-stage ‘recruits’ from the estuary. Eel density in the pre-impounded zone increased from 0.5 per 100 m^2 before barrier removal to 32.5 per 100 m^2 five months after removal. By 17 months after barrier removal there was no difference in eel density across the six sections sampled. Although resident stream fishes such as bullhead (*Cottus gobio* species complex, protected under EU Habitats and Species Directive) were abundant in middle and upper-stream sections, brown trout (*Salmo trutta,* a listed species for biodiversity conservation in England/Wales) density remained low during the study and recruitment was poor. This suggests that although colonisation access for anadromous trout was available, habitat upstream may have been unsuitable for reproduction, indicating wider catchment management is required to complement connectivity restoration. Slower recolonization by trout might also not be reflected in a short study such as this one. Findings of this study suggest that tidal barrier removal is an effective method of restoring lotic habitats and connectivity, and can be beneficial for resident and migratory fishes including those of conservation importance (e.g. European eel) in coastal streams.

4.1 Introduction

In-stream obstacles such as dams, weirs and culverts fragment rivers by interrupting longitudinal connectivity and altering habitat (Nilsson *et al.*, 2005; Sun *et al.*, 2020, also see Chapter 3), having major effects on the biodiversity and functioning of river ecosystems (Bunn and Arthington, 2002; Pringle, 2003; Galib *et al.*, 2018, also see Chapter 1). These obstacles frequently impact the dispersal and migration of fish species, and can result in population decline and biodiversity reduction (Lucas and Baras, 2001; Gehrke *et al*., 2002; Katano *et al*., 2006; Mueller *et al*., 2011, also see Chapter 2). The flow-impounding effects of river barriers result in alteration to slower, deeper, fine-sediment dominated habitat immediately upstream, especially in low-gradient reaches, with resultant effects on the biota (Boon, 1988; Mueller *et al.*, 2011; Birnie-Gauvin *et al.*, 2017a). Barriers close to the sea can have disproportionate effects on diadromous fish species distribution in rivers by limiting access to suitable habitat upstream (Kemp and O'Hanley, 2010; Nunn and Cowx, 2012; Harris, 2016).

One such species is the European eel (*Anguilla anguilla*), the abundance of which has decreased greatly since the early 1980s (Dekker, 2003; Henderson *et al.*, 2012; Jacoby *et al.*, 2015). Recruitment of glass eel (the transparent juvenile stage) has reduced by more than 90% and the population of silver eel (migrating to sea) has reduced by more than 50% (Piper *et al.*, 2013; Jacoby and Gollock, 2014). In this case, the term "recruitment" as used in this chapter, means the arrival of the early-life stage glass eel to estuaries (Cresci *et al*., 2019), rather than changes in local abundance due to in situ reproduction and subsequent transition through life stages, relative to mortality. Due to its rate of decline, this species has been classified as 'Critically Endangered' in the International Union for Conservation of Nature (IUCN) global Red List (Jacoby and Gollock, 2014). Under the Water Framework Directive (WFD), EU countries are required to provide free migration of fishes (European Commission, 2003, also see Chapter 1), which is a particularly relevant policy tool for supporting the recovery of European eel, as well as for other migratory fish species such as brown trout (*Salmo trutta*). The European Commission also initiated an Eel Recovery Plan (Council Regulation No 1100/2007) to ensure sustainable levels of adult eel abundance and glass eel recruitment across the European Union (European

Commission, 2007a). Through this, EU states are required to develop Eel Management Plans across River Basin Districts. Also in 2007 (ratified 2009), European eel was listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), allowing export only without detriment to the species survival. As a result, since December 2010, all commercial trade of European eel to and from the EU has been banned (Musing *et al.*, 2018).

The reasons for the decline in recruitment of European eel are still not fully understood, with various threats ranging from overexploitation to climate change, but are mirrored by several other *Anguilla* species (Jacoby *et al.*, 2015). However, the occurrence of in-stream barriers restricting access to juvenile habitat is considered to be one of the major threats to the European eel population (Dekker, 2003; Piper *et al.*, 2013). It is also a threat that can be responded to, through river restoration. In England and Wales, eels are widely distributed throughout the river system. However, catches of eel in England and Wales have largely declined since the late 1970s (Defra, 2010a). For example, eel abundance in Bridgwater Bay, Somerset, England reduced 99% between 1980 and 2009 (Henderson *et al.*, 2012). However, there had not been a concerted effort to record changes in eel abundance until recently, with most good English datasets being on the Rivers Severn and Thames. There is no fishery-independent monitoring of glass eel recruitment in the Northumbria River Basin and surveys of yellow eel distribution have been poorly standardised (Defra, 2010b). Results of electro-fishing surveys revealed that sites with eel presence steadily declined in the Rivers Aln, Coquet, Tyne and Wear between 1999 and 2005 (Defra, 2010b).

The upstream migration of juvenile European eel can last several years during which time they may migrate hundreds of kilometres and grow to ~40 cm, although a proportion never enter fresh water (Lucas and Baras, 2001). Barriers such as weirs, dams and sluices limit their upstream migration, restricting access to suitable nursery habitat upstream (Mouton *et al.*, 2011; Tamario *et al.*, 2019). Although juvenile eel, especially those smaller than 10 cm, can climb and crawl on wet and rough surfaces (Porcher, 2002; Watz *et al.*, 2019), only a small proportion of them may manage to pass barriers (White and Knights, 1997;

Tamario *et al.*, 2019). In-stream barriers and associated engineering infrastructure can reduce survival and delay the downstream migration of the maturing silver eel stage (Behrmann-Godel and Eckmann, 2003; Calles *et al.*, 2010; Piper *et al.*, 2013) before migration to oceanic spawning grounds. For all diadromous species, enabling their bidirectional migration in rivers is crucial (Calles and Greenberg, 2009).

Although a variety of fishway designs exist to facilitate upstream and/or downstream migration (Silva *et al.*, 2018), their efficacy for many species can be low (Bunt *et al.*, 2012). Eel-specific fishways pass a proportion of juvenile eel (Environment Agency, 2011; Watz *et al.*, 2019) but are unsuitable for most other species. Tide flaps and management of sluices can also be used to support the passage of eels in tidal reaches (Environment Agency, 2011; Wright *et al.*, 2015; Guiot *et al.*, 2020). Unlike these mitigative measures, barrier removal reinstates hydrologic connectivity, more natural habitat, sediment transport, and free movement of aquatic biota (Roni *et al.*, 2008; Kemp and O'Hanley, 2010; O'Hanley, 2011, also see Chapter 1). Removal of redundant barriers is increasingly used as a river management and conservation tool in many countries (Birnie-Gauvin *et al.*, 2017c; Silva *et al.*, 2018). Several studies have measured the effects of barrier removal on geomorphological and ecological responses in rivers (Pizzuto, 2002; Doyle *et al.*, 2005; Chang *et al.*, 2017; Clark *et al.*, 2020). However, for barriers that occur in tidal reaches, the recovery of fish communities in response to barrier removal is still poorly known (but see Souder *et al.*, 2018). The tidal sections of rivers are characterised by large fluxes of nutrients, sediment and organisms, between marine and freshwater environments (Levin *et al.*, 2001) and barriers that interrupt tidal reaches can dramatically alter these. Free-flowing rivers provide a range of physical habitats which are important for supporting the fish populations (Brink *et al.*, 2018). Therefore, removal of tidal / tidal-limit barriers may be hypothesised to have a rapid effect on changes in local habitat and fish community through reinstating sediment transport, bidirectional flow and facilitating fish dispersal and migration. In particular, removal of such barriers is predicted to benefit the migration and production of species such as European eel. Moreover, aquatic invertebrates are important food sources for many fish species (e.g. European eel, brown trout and bullhead *Cottus gobio* species complex) and changes in aquatic habitats due to impounding effects may alter invertebrate assemblages (Vinson, 2001), which could affect the diversity and abundance of fishes.

In this study, the changes in aquatic habitat, fish abundance, and fish and benthic invertebrate communities were measured in response to the removal of a tidal weir in a small stream of the River Tees, northeast England. A before-after methodology was used, and particularly focused on the recolonisation of European eel in the stream. Although the primary action was removal of a tidal-limit weir, the study operated over multiple sites along the entire stream catchment to determine wider-scale as well as local effects. It was hypothesised that the tidal barrier removal would result in the change of habitat from impounded, lentic water to more diverse habitat, with associated rapid change in the fish community in the formally impounded zone and benefit the recruitment of diadromous fishes in the stream.

4.2 Methods

4.2.1 Study site

Claxton Beck, northeast England, is a low-gradient stream which joins Greatham Creek within the intertidal zone of the River Tees (Figure 4.1) downstream of the Tees Barrage. Claxton Beck is a small watercourse (1 – 4 m wide at the natural tidal limit, after barrier removal) that rises at an altitude of 126 m. Claxton Beck and its upstream reach, North Burn, drain an area of 41 km^2 before joining Greatham Creek. The latter is located in an area surrounded by wet pasture and mudflats. Cloff Bridge weir, a barrier located at the head of tide, (54°37'39.2"N 1°15'14.5"W) was built around 1910 to prevent tidal intrusion and so enable abstraction of fresh water, from above the weir, to a nearby brickworks (now defunct). The weir was a 2.4-m high concrete structure (Figure 4.2) that was impassable to most fish species under most conditions, and a major obstruction to eel. Throughout much of the 20th Century the Tees estuary was heavily polluted by industrial and urban waste, with an impoverished fish community, but the estuary became cleaner from the 1980s onwards, enabling progressive recovery of the fish community and recolonisation of suitable habitat (see Chapter 2 for a fuller discussion of the decline and recovery of the Tees).

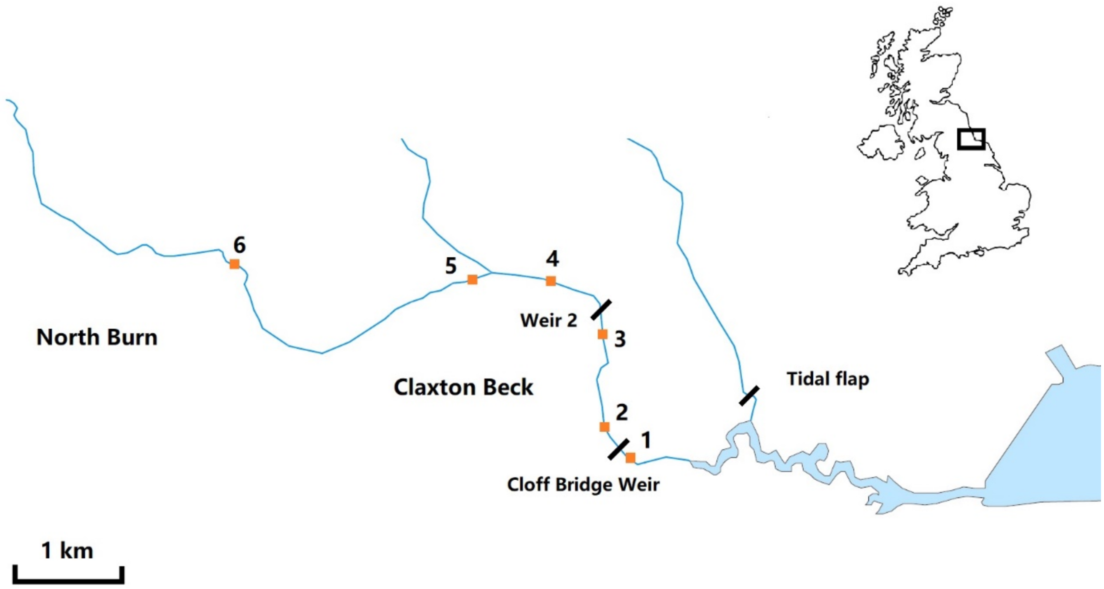


Figure 4.1 Claxton Beck catchment and the midpoint of each fish sampling section. Sampling only occurred on Claxton Beck / North Burn.

Figure 4.2 Cloff Bridge weir on Claxton Beck before removal (a), immediately after removal on 30 April 2018 (b), five months after the weir removal (c) and 17 months after the weir removal (d). Photographs were taken under base flow conditions.

Prior to the agricultural and Industrial Revolutions, small streams entering northern English estuaries, such as Claxton Beck entering the Tees estuary, would likely have been populated by a fish community comprising diadromous migratory fishes, especially brown trout, European eel, European flounder (*Platichthys flesus*), other euryhaline species (those tolerant to salinity fluctuations such as several goby and stickleback species) and small, resident species such as stone loach (*Barbatula barbatula*) (Wheeler, 1969). In small, lowland Danish coastal streams, similar in climate and natural hydromorphology to Claxton Beck, anadromous brown trout are often the dominant species (Birnie-Gauvin *et al.*, 2018). However, agricultural intensification, land drainage, stream straightening and pollution degraded the habitat of many lowland streams across England, including Claxton Beck, so although Claxton Beck probably once contained a substantial brown trout population, it was almost extirpated. The Environment Agency (EA) stocked North Burn with brown trout fry in 1997 but they did not perpetuate at the sites stocked (R. Jenkins, EA, pers. comm. see also Figure 2.47). In recent years small numbers of adult sea trout (*S. trutta*) have been observed in the reach downstream of Cloff Bridge weir during the autumn breeding season (B. Lamb, Tees Rivers Trust, pers. comm.).

Due to Cloff Bridge weir's impounding effects, the upstream reach was dominated by a 480-m long, uniformly deep (~ 1 m), slow glide, with fine sediment on the bed. A small scour pool with industrial rubble and gravel had developed immediately downstream of the weir. The weir formed an artificial tidal limit, and within a few hundred metres downstream of the weir, the channel became progressively more characteristic of a tidal creek, dominated by tidally transported soft sediment, with exposed mud banks and marginal reeds (*Phragmites australis*). From WFD monitoring, in the upstream freshwater reach, the ecological status of the fish community at Claxton Beck was classified as "bad" between 2013 and 2016 by the EA (Environment Agency, 2020a). Cloff Bridge weir was considered to be the main reason of WFD failure for fish. Another 1-m high weir is located 1.5 km upstream of Cloff Bridge weir and 0.2 km downstream of a major road bridge that crosses Claxton Beck.

In order to provide free passage for fish and help restore the upstream river habitat, by reinstating flow and sediment connectivity, Cloff Bridge weir was removed by the Tees Rivers Trust on 30 April 2018 (Figure 4.2). Funding for this action was provided by EDF Energy as a mitigation, by agreement with the Environment Agency, for the impact of eel impingement at the nearby Hartlepool Nuclear Power station that abstracts cooling water from the Tees estuary. The unnamed weir 1.5 km upstream of Cloff Bridge could not be removed due to concerns over potential stream bed erosion actions on the road bridge upstream. Therefore, a wooden pool fish pass was installed by Tees Rivers Trust on the middle of the second weir in June 2018. The slope of the fish pass is 30 degrees, it has nine 0.1-m high pools and a width of 0.5 m. In addition, a bristle elver pass was installed on the left side of the pool fish pass in September 2018.

4.2.2 Experimental approach

Because environmental conditions, especially flow, vary seasonally and could be expected to alter habitats, especially after weir removal, sampling of habitat (especially around Cloff Bridge weir) and biota was carried out twice a year, in autumn and spring. Samples were taken on five occasions; in autumn 2017 and spring 2018, prior to removal of Cloff Bridge weir, and in autumn 2018, spring 2019 and autumn 2019 after barrier removal. Spring samples were carried out in April and autumn samples in late September and early October.

4.2.3 Habitat measurements

In order to assess river habitat close to Cloff Bridge weir before and after barrier removal, a habitat survey was performed during seasonal base flow conditions. Hydromorphological characteristics comprising wetted width, depth (at 25%, 50% and 75% of wetted width) and flow velocity (at 50% depth and 25%, 50% and 75% of wetted width) were measured every 12 m in the upstream impounded section (length, 480 m) and downstream tidal section (length, 204 m). This 12-m distance repeat reflected the typical field of view of photographs made looking up the channel, that were made before and after barrier removal, through these upstream and downstream zones. These measurements were made 7 months before (Sep 2017) and 5 months (Sep 2018), 12 months (Apr 2019) and 17 months (Sep 2019) after weir removal. Sampling could not be carried out in April 2018 due to prioritisation of biotic sampling during the only period of low flows that month. Sampling in the tidal section was carried out close to low tide. Dominant habitat types (riffle, glide, pool) and substrate types (sand, silt, gravel etc.) in each 12-m section were recorded. The river bed substrate composition in each 12-m section was visually and manually assessed, using an approximation to the Wentworth scale: boulder (>256 mm), cobble (64–256 mm), gravel (2–64 mm), sand (0.06–2 mm) and silt (<0.06 mm) (Wentworth, 1922; Environment Agency, 2003). An additional substrate category "earth" was used to describe compacted soil (inorganic and organic materials) that formed submerged banks and, in some areas part of the stream bed, particularly within the inundated impounded reach.

A more detailed survey grid of water depth and flow velocity was carried out in zones stretching 20 m upstream and 40 m downstream of the weir's midpoint. These data were used for generating 2D graphs, to visualize habitat changes post-weir removal. Both characteristics were measured on a 1-m mesh. If the wetted width was less than 3 m, then measurements were taken at 25%, 50% and 75% width positions. An electromagnetic flow meter (Valeport 801) was used to take flow velocity measurements except for in a 10-m long section affected by electromagnetic interference from high voltage electricity transmission pylons, where an analogue Hydro-prop Impeller flow meter was employed.

4.2.4 Sample sections for biota surveys

Six sampling sections, each 300-m long, were chosen in which to sample biota (Figure 4.1). It was feasible to sample only one section downstream of Cloff Bridge weir due to the deep, soft mud further downstream. In Section 1, located immediately downstream of the weir (Figure 4.1), the tide mark was approximately 1 m high on the banks. The riparian zone of Section 1 is mostly semi-improved grassland. Land use adjacent to Section 1 is pasture and arable land on the left bank and semi-natural parkland on the right bank. Downstream of Section 1, the riparian zone is dominated by common reed. Section 2 was located immediately upstream of the weir, within the impounded zone and Section 3 was located nearly 700 m upstream of Section 2, downstream of the second weir. Sections 4–6 were located upstream of the second weir (Figure 4.1). The riparian zone of Sections 2–6 mostly consists of broadleaf trees such as sycamore (*Acer pseudoplatanus*) and common alder (*Alnus glutinosa*) along with some tall herbs such as nettle (*Urtica dioica*) and butterbur (*Petasites hybridus*). The predominant land use adjacent to Sections 2–5 is mixed agricultural land. For Section 6, the land use is semi-improved grassland on the left side and broadleaf woodland on the right side. Apart from Section 2 which, prior to weir removal was an impounded area, the remaining sampling sections contained multiple habitat types (riffle, glide and pool). Because the second weir has not been removed, the removal of Cloff Bridge weir is unlikely to have had any impacts on river habitat upstream of the second weir. Initially, Section 3 was positioned 700 m upstream of Section 2, but due to difficulties with land access permission there after summer 2018, the sampling section was moved 500 m further upstream until the end of the study. The new Section 3 had similar river habitat compared with the original location, and the post-weir removal fish population surveys were all conducted in the new Section 3.

4.2.5 Fish community sampling

Fish were sampled by electrofishing using wading with a single anode, operated with a bankside generator and control box (Honda EU10i, Electracatch WFC1, ~200 V). For Section 1 in the tidal reach (Figure 4.1), sampling was carried out close to low tide, when depth and conductivity were lowest (always <1 ppt salinity). Although single-funnel, 5-mm mesh, baited traps were trialled as another method of sampling fish, these were ineffective and their use was discontinued. Six 20-m long, full channel-width sample replicates, targeting a mixture of habitat types, approximately proportionally to their availability were spread along each 300-m sample section.

The three-pass electrofishing ‘depletion’ method (Reynolds and Kolz, 2013) was carried out for each 20-m sample length, using 4-mm mesh stopnets to delimit the fished section. After the first and second rounds of electro-fishing, approximately 30 minutes was given to let the sediment settle down and allow fish to return to their activities, to generate relatively equal catchability between rounds (based upon experience and catchability measurements at local stream sites). Fish removed from each pass were kept in separate aerated containers, after which the catches were processed separately. Fish were identified and measured for total length. If more than 50 fish of a species were caught at a site, then 50 per species were randomly selected and measured, and the remainder counted. Processed fish were released back to the capture location. All fish sampling was carried out under permit, issued by the Environment Agency.

4.2.6 Invertebrate sampling

Four sections, were chosen for conducting benthic macroinvertebrate sampling: Sections 1–4. Sections 5 and 6 were not sampled because it was expected that no rapid changes in invertebrate communities would occurred in the sections located furthest upstream, since none of the Tees brackish-water invertebrates are capable of colonising freshwater. Three sampling sites were sampled in each section, and each site was surveyed twice per year, once in spring and once in autumn. All in-stream habitats were kick sampled in proportion to their occurrence, for a total of three minutes using a handnet with 1-mm mesh, plus one minute hand searching. At sites with little flow, material suspended by kick sampling was washed into the net by generating flow with a hand or foot. After sampling, all invertebrates were stored in 70% ethanol and identified to family level in the laboratory using a binocular microscope and standard literature (e.g. Pawley, 2011). In two cases the level of taxonomic resolution was not to family level: Oligochaeta and Mysidacea.

4.2.7 Data analysis

Before analysis, data were checked for normality using conventional tests, and necessary transformations were applied when needed. For habitat metrics, pairwise Permutational multivariate analysis of variance (PERMANOVA) from the ‘RVAideMemoire’ package (Hervé, 2020) was applied to analyse whether the habitat types, substrate types and hydromorphology (water depth, flow velocity and wetted width) differed between the downstream section (Section 1) and upstream impounded section (Section 2) (Chang *et al.*, 2017). All habitat data were log(x+1) transformed before conducting analyses. Before and after changes in water depth and flow velocity immediately upstream and downstream of the weir were visualised using Iric (version. 2.3) (Nelson *et al.*, 2016).

Fish densities per site and species were calculated according to Carle and Strub's K-pass removal method, by using the R (version 3.6.1) package 'FSA' (Ogle, 2020). Total fish densities were calculated by summing densities of individual species per site (from Carl and Strub estimates), taking account of differing species cathabilities in doing so. Fish densities and invertebrate relative abundance data were fourth-root transformed, to meet assumptions of normality before conducting the following analysis (Boys *et al.*, 2012). PERMANOVA was used to determine changes in the fish and invertebrate communities after weir removal, using the R 'Vegan' package (Oksanen *et al.*, 2019). In order to create a balanced design to perform PERMANOVA, the surveys were split into three periods, each comprising a spring survey and an autumn survey (Period 1: autumn 2017, spring 2018; Period 2: autumn 2018, spring 2019; Period 3: spring 2019, autumn 2019). Similarity percentage (SIMPER) analysis, based on the decomposition of Bray-Curtis dissimilarity index (Clarke, 1993), was used to identify the contribution of individual species to the overall fish community at each section. Linear Mixed-Effects Models (LMMs) were performed to analyse the changes in fish abundance using the 'lme4' and 'lmerTest' package (Kuznetsova *et al.*, 2017). Tukey's multiple comparison test was performed to analyse the differences in total fish abundance (all fish species combined) and eel abundance between study sections, using the 'multcomp' package (Hothor *et al.*, 2020). Sites (nested within sections) and seasons (nested within sampling years) were used as random factors when performing both analyses. To visualize the spatial and temporal differences in fish communities, a Non-metric multidimensional scaling (NMDS) (Kruskal and Wish, 1978) ordination plot was generated using the 'metaMDS' function of the 'vegan' package. Examples of R-scripts are presented in Appendix II.

Invertebrate communities are good indicators of watercourse pressures (e.g. pollution), and they are frequently used in assessing the level of general degradation (UK Technical Advisory Group, 2014). The WHPT ASPT (Whalley, Hawkes, Paisley & Trigg – Average Score Per Taxon) was applied as an abundance weighted metric (UKTAG, 2014) for assessing responses of the invertebrate community across stream sections before and after barrier removal. The ASPT at each section was also analysed by using LMM.

4.3 Results

4.3.1 Aquatic habitat pre- and post-barrier removal

Photographs of example sections of habitat upstream along the 480-m upstream section ad 204-m downstream section before and after barrier removal are presented in Appendix II (Downstream: Figure S4.1, S4.2; Upstream: Figure S4.3, S4.4). Before barrier removal, Section 2 was impounded and dominated by a deep, very slow glide (Table 4.1). Substrate in the impounded section was mostly composed of sand (mean ± SD, 68.7 ± 33.7 %), along with some exposed earth (mean ± SD, 12.6 ± 28.2 %) close to the water's edge, and silt (11.4 ± 23.3 %) accumulated on the upstream side of the weir (Table 4.1). Downstream, and before the weir's removal, the stream was shallower (mean ± SD, 28.2 ± 9.0 cm) and narrower (mean ± SD, 4.22 ± 1.62 m). The tidal stream section exhibited a more natural form with faster flow (0.08 ± 0.04 m s^{-1}) and glide, riffle and pool habitats at low tide (Table 4.1). Mud (mean ± SD, 46.7 ± 25.3 %) and sand (mean ± SD, 23.6 ± 13.5 %) formed the majority of the bottom substrates, but gravel and boulder occurred intermittently (Table 4.1).

Although the whole weir was removed, a steep riffle remained at its former position (Figure 4.2) and the tidal limit remained in the vicinity of the former weir's position for the duration of the study. The riparian vegetation and river bank canopy in the former impounded zone and downstream tidal zone were not affected by barrier removal. Bed substrates, habitat types and hydromorphology exhibited dramatic changes in Section 2 within the first 5 months after barrier removal (PERMANOVA, $P < 0.05$ in all cases; Table 4.2). Section 2 became shallower and narrower, with faster flow (Figure 4.3). Large volumes of fine sediment were washed to the downstream section; the proportion of bed in Section 2, covered by sand decreased from 68.7 to 20.4% five months after barrier removal, then slightly increased to 31.4%, after 12 months (Table 4.1). After the sand was washed away, it exposed underlying compacted earth of the channel bed, and this became the dominant substrate in Section 2, increasing to 60.7% coverage five months after barrier removal, then slightly decreasing to 44.5%, 17 months after barrier removal (Table 4.1).

The overall upstream substrate composition appeared stable at 17 months after the barrier removal (Table 4.3; PERMANOVA pairwise post hoc, $P = 0.078$). A few riffles and pools were formed in the previously impounded section (Table 4.1) and caused significant changes in habitat type occurrence at 5 months post-removal (PERMANOVA pairwise post hoc, $P = 0.002$) with no further changes at 12 and 17 months post-removal (PERMANOVA pairwise post hoc, $P > 0.05$ in both cases). Similar to the habitat factors, water depth, wet width and flow velocity all changed within 5 months post-removal (PERMANOVA pairwise post hoc, $P = 0.002$), then these factors became stable and showed no further significant changes (Table 4.3).

Table 4.1 Percentage occurrence of habitat types (mean and SD), hydromorphology and substrate composition (percentage of bed) in the upstream (u/s; length, 480 m) and downstream (d/s; length, 204 m) study reaches after different periods, 7 months before weir removal (Sept 2017); 5 months post removal (Sept 2018); 12 months post removal (Apr 2019); 17 months post removal (Sept 2019). “Earth” refers to compacted bank soil in inundated areas.

Habitat factors			7 months before	5 months after	12 months after	17 months after
Habitat types	Riffle %)	u/s	0.0±0.0	13.2±30.4	17.3±31.5	11.4±22.9
		d/s	16.7±26.3	28.1±38.7	31.7±40.3	16.7±28.0
	Glide (%)	u/s	100.0±0.0	58.0±41.4	68.9±39.0	76.2±31.8
		d/s	59.2±39.1	60.0±42.8	60.0±40.6	67.2±34.0
	Pool (%)	u/s	0.0±0.0	28.8±38.3	13.8±27.5	12.4±23.9
		d/s	24.2±38.2	11.9±29.6	8.33±21.15	16.1±30.0
Hydro-morphology	Depth (cm)	u/s	104.5±23.1	46.5±27.3	40.1±22.6	41.4±23.0
		d/s	28.2±9.0	22.9±17.2	23.5±13.4	26.6±16.2
	Wet width (m)	u/s	5.98±1.66	3.25±1.18	3.28±1.12	3.35±1.13
		d/s	4.22±1.62	3.66±1.43	3.83±0.88	3.71±0.62
	Velocity (m s^{-1})	u/s	0.01±0.01	0.04±0.06	0.04±0.04	0.03±0.05
		d/s	0.08±0.04	0.07±0.09	0.10±0.11	0.06±0.08
Substrate composit-eon	Silt (%)	u/s	11.4±23.3	12.6±23.2	11.6±22.3	6.5±16.8
		d/s	46.7±25.3	72.3±31.4	27.2±20.9	32.5±32.2
	Sand (%)	u/s	68.7±33.9	20.4±29.0	31.4±26.2	38.8±31.8
		d/s	23.6±13.5	5.8±15.1	34.7±26.1	36.7±28.1
	Gravel (%)	u/s	0.7±4.1	3.0±10.4	4.9±11.1	0.8±2.7
		d/s	15.8±14.1	8.9±13.5	16.9±24.0	12.8±22.3
	Cobble / Boulder (%)	u/s	6.8±15.2	3.4±7.2	3.8±7.1	9.5±19.7
		d/s	13.9±22.8	13.8±17.2	13.9±21.6	14.2±19.9
	Earth (%)	u/s	12.6±28.2	60.7±38.6	48.4±31.8	44.5±34.3
		d/s	0.0±0.0	0.0±0.0	7.8±18.7	3.9±12.1

Table 4.2 Pairwise PERMANOVA comparing river habitat factors before (7 months before removal) and after (5, 12, 17 months post removal) the weir removal. Significant values in bold.

	Habitat factors	Mean square	*df*	*F*	*P*
Downstream	Substrates	7.007	1,70	3.55	**0.008**
	Habitat types	0.617	1,70	0.33	0.782
	Hydromorphology	0.075	1,70	0.78	0.406
Upstream	Substrates	17.555	1,146	7.94	**0.001**
	Habitat types	6.211	1,146	4.60	**0.008**
	Hydromorphology	4.734	1,146	44.27	**0.001**

Table 4.3 Post hoc tests following pairwise PERMANOVA showing the temporal variation in river habitat during sampling periods. Sep 2017: 7 months before weir removal; Sep 2018: 5 months after weir removal; Apr 2019: 12 months after weir removal; Apr 2019: 17 months after weir removal. Pillai post-hoc test applied. Significant values in bold.

	Habitat factors	Periods	*P*		
Downstream	Substrates		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	**0.003**	-	-
		Apr 2019	**0.016**	**0.003**	-
		Sep 2019	**0.003**	**0.004**	0.697
	Habitat types		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	0.78	-	-
		Apr 2019	0.78	0.91	-
		Sep 2019	0.91	0.78	0.78
	Hydromorphology		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	0.18	-	-
		Apr 2019	0.40	0.58	-
		Sep 2019	0.13	0.92	0.53
Upstream	Substrates		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	**0.002**	-	-
		Apr 2019	**0.002**	**0.042**	-
		Sep 2019	**0.002**	**0.033**	0.078
	Habitat types		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	**0.002**	-	-
		Apr 2019	**0.002**	0.337	-
		Sep 2019	**0.002**	0.223	0.693
	Hydromorphology		Sep 2017	Sep 2018	Apr 2019
		Sep 2018	**0.002**	-	-
		Apr 2019	**0.002**	0.897	-
		Sep 2019	**0.002**	0.792	0.897

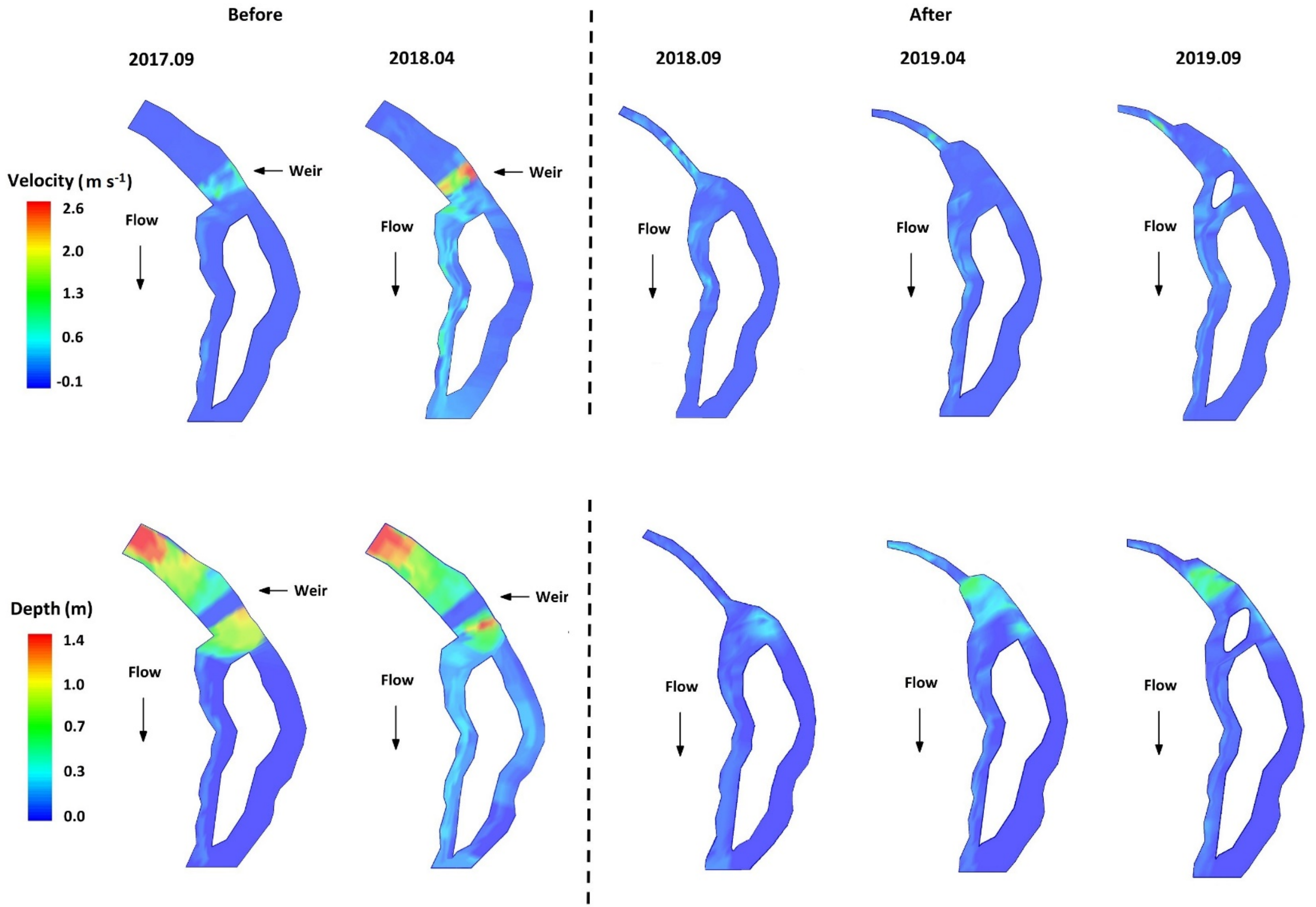


Figure 4.3 Flow velocity and water depth before and after the weir removal (upper panel: Velocity; lower panel: Depth).

In the downstream reach, Section 1, the bed substrate composition changed after weir removal (PERMANOVA pairwise post hoc, $P = 0.002$), with much of the bed covered by 10-cm thick silt. The proportion of substrate classed as silt increased from 46.7 to 72.3% at 5 months post-removal (Table 4.1). Most surface silt was washed further downstream after several winter high-flow events by 12 months post-removal, and the silt proportion in Section 1 reduced to 27.2% (Table 4.1). Meanwhile, the proportion of sand increased from 5.8 to 34.7%. The overall bottom substrates showed no further change by 17 months post-removal (PERMANOVA pairwise post hoc, $P = 0.697$). The habitat factors were not affected by the barrier removal in the downstream reach through the study periods (PERMANOVA pairwise post hoc, $P > 0.05$ in both cases).

4.3.2 Fish abundance and fish community pre- and post-barrier removal

Before barrier removal, eight fish species were captured during the electrofishing surveys (Figure 4.4). European flounder, nine-spined stickleback (*Pungitius pungitius*) and common goby (*Pomatoschistus microps*) were only captured in Section 1. Section 1 was dominated by flounder in spring and by European eel in autumn. The predominant species in Section 2 was three-spined stickleback (*Gasterosteus aculeatus*), and sites further upstream (Section 3–6) were dominated by bullhead (*Cottus perifretum*, part of the *C. gobio* species complex *sensu* (Freyhof *et al.*, 2005)). Before barrier removal, among all sampled sections, total fish density in Section 1 was significantly higher than in all upstream sections in autumn (Figure 4.5; Table 4.4; LMM pairwise post hoc, $P < 0.05$ in all cases) and fish density in Section 2 was significantly lower than in all other sections in spring (Figure 4.5; Table 4.4; LMM pairwise post hoc, $P < 0.05$ in all cases).

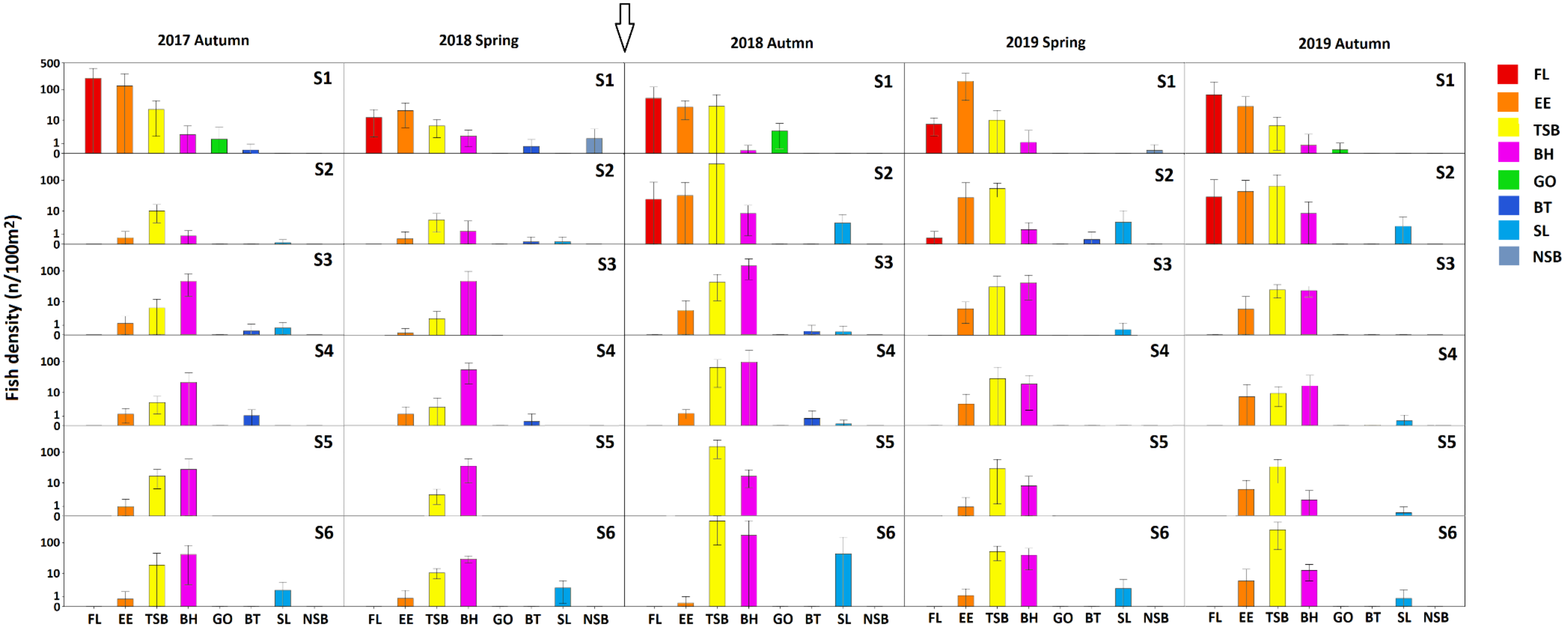


Figure 4.4 Mean fish densities (per 100 m^2) of each species in each section before and after the barrier removal. Error bar: 95% confidence interval. FL: European flounder, EE: European eel, TSB: three-spined stickleback, BH: European bullhead, GO: common goby, BT: brown trout, SL: stone loach, NSB: nine-spined stickleback. The arrow signifies when the barrier was removed. Section numbers are ordered from downstream to upstream, with Section 1 being downstream of the barrier location and Section 2 the impounded zone prior to barrier removal.

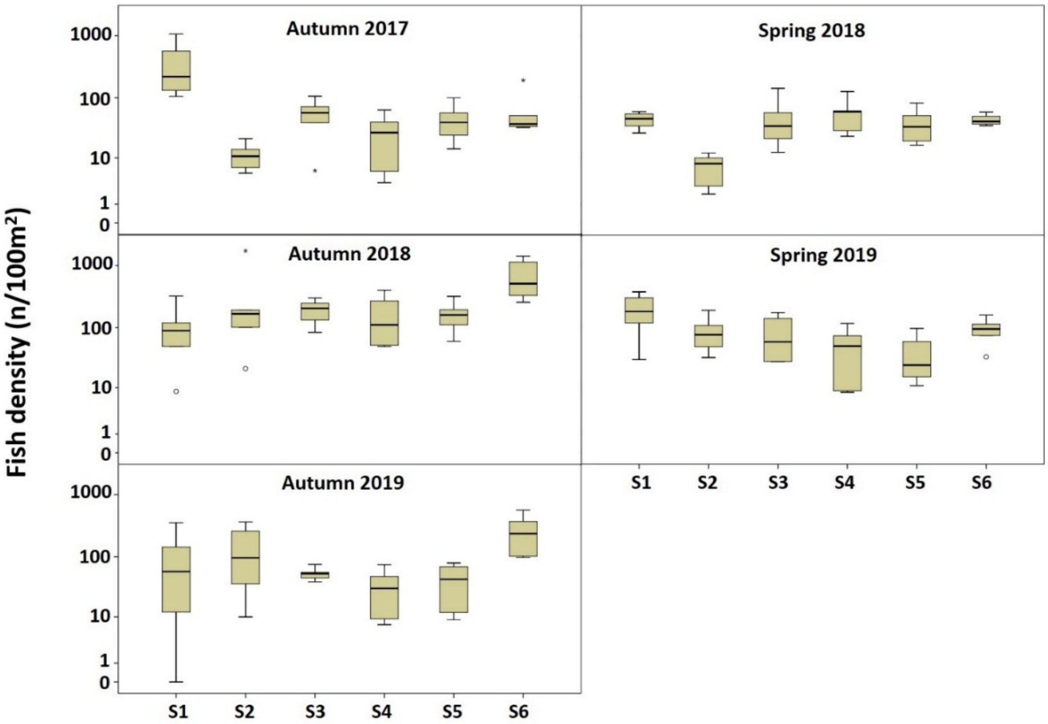


Figure 4.5 Box plots showing median (with quartiles, ranges and outliers) total fish densities (per 100 m^2) at each section before and after the weir removal (removed after spring 2018 sampling). Section 1 is the furthest downstream site, in the tidal zone, Sections 2-6 are non-tidal, with Section 6 the furthest upstream.

Table 4.4 Pairwise comparisons of total fish density in each section during the survey periods. Sections 1 and 2 are immediately downstream and upstream of the barrier position; section numbers increase from downstream to upstream. Tukey's multiple comparison test applied. Significant values are shown in bold.

Periods	Before				After					
Season	autumn 2017		spring 2018		autumn 2018		spring 2019		autumn 2019	
Section	*Z*	*P*	*Z*	*P*	*Z*	*P*	*Z*	*P*	*Z*	*P*
2 - 1	-6.87	<**0.001**	-4.81	<**0.001**	1.55	0.632	-1.95	0.371	1.29	0.789
3 - 1	-4.50	<**0.001**	-0.23	1.000	1.36	0.750	-2.30	0.192	0.34	0.999
4 - 1	-5.89	<**0.001**	0.67	0.986	0.77	0.972	-3.49	**0.006**	-0.52	0.995
5 - 1	-4.78	<**0.001**	-0.63	0.989	1.02	0.910	-3.91	**0.001**	-0.21	1.000
6 - 1	-4.27	<**0.001**	0.01	1	3.91	**0.001**	-1.68	0.544	2.83	0.052
3 - 2	2.37	0.167	4.57	<**0.001**	-0.19	1.000	-0.35	0.999	-0.95	0.934
4 - 2	0.98	0.926	5.47	<**0.001**	-0.78	0.972	-1.54	0.640	-1.81	0.457
5 - 2	2.09	0.291	4.18	<**0.001**	-0.53	0.995	-1.96	0.366	-1.50	0.666
6 - 2	2.60	0.097	4.81	<**0.001**	2.37	0.169	0.27	1.000	1.54	0.638
4 - 3	-1.40	0.730	0.90	0.947	-0.59	0.992	-1.19	0.844	-0.86	0.955
5 - 3	-0.28	1.000	-0.39	0.999	-0.34	0.999	-1.61	0.594	-0.55	0.994
6 - 3	0.23	1.000	0.24	1.000	2.55	0.109	0.62	0.989	2.49	0.127
5 - 4	1.12	0.875	-1.29	0.789	0.25	1.000	-0.42	0.998	0.32	1.000
6 - 4	1.63	0.580	-0.66	0.986	3.14	**0.021**	1.81	0.461	3.36	**0.010**
6 - 5	0.51	0.996	0.63	0.989	2.89	**0.044**	2.23	0.224	3.04	**0.029**

After barrier removal, the predominant species of Sections 1–6 remained similar, but eel became relatively more abundant further upstream than previously (Figure 4.4). The overall fish densities across all sections exhibited a significant increase in density after barrier removal (Figure 4.5, Table 4.5; LMM, $F_{1,143} = 14.154$, $P < 0.001$). Five months after barrier removal, the fish abundance in Section 2 had dramatically increased, and there was no significant difference in total fish density between Section 2 and all other sections (LMM, $P > 0.05$ in all cases).

Table 4.5 Linear mixed-effects model (LMM) results showing changes in total fish abundance in each section before, compared to after, barrier removal (Before: autumn 2017 and spring 2018; After: autumn 2018, spring 2019 and autumn 2019). Season and site were used as random factors in the analysis. Significant values in bold.

Section	Fish Species	Mean square	*df*	*F*	*P*	*Trend*
All combined	Overall fish	14.154	1,143	21.584	<**0.001**	↑
1	Overall fish	0.697	1,23	0.620	0.439	**-**
2	Overall fish	17.966	1,23	30.487	<**0.001**	↑
3	Overall fish	1.879	1,23	6.281	**0.020**	↑
4	Overall fish	0.650	1,23	2.127	0.158	**-**
5	Overall fish	0.432	1,22	1.330	0.261	**-**
6	Overall fish	10.476	1,27	19.904	<**0.001**	↑

The fish communities differed significantly between Sections 1, 2 and 3 before the barrier removal (PERMANOVA, $P < 0.05$ in all cases; Table 4.6). The fish communities in Sections 1, 2 and 3 changed after barrier removal (Figure 4.6, Table 4.7; PERMANOVA, $P < 0.05$ in all cases) and remained different from each other after barrier removal in both P2 (autumn 2018 – spring 2019) and P3 (spring 2019 – autumn 2019) (PERMANOVA, $P < 0.05$ in all cases). For Section 1, SIMPER showed eel and flounder contributed over 80% of the change in fish assemblages after barrier removal, in both P2 and P3. For Section 2, both three-spined stickleback and eel abundance increased significantly after barrier removal (LMM, stickleback: $F_{1,28} = 21.599$, $P < 0.001$; eel: $F_{1,23} = 16.782$, $P < 0.001$), and these two species contributed over 80% of the dissimilarity in fish assemblages after the barrier removal. There was little change further upstream. Eel contributed less to fish communities change at S3 – S6.

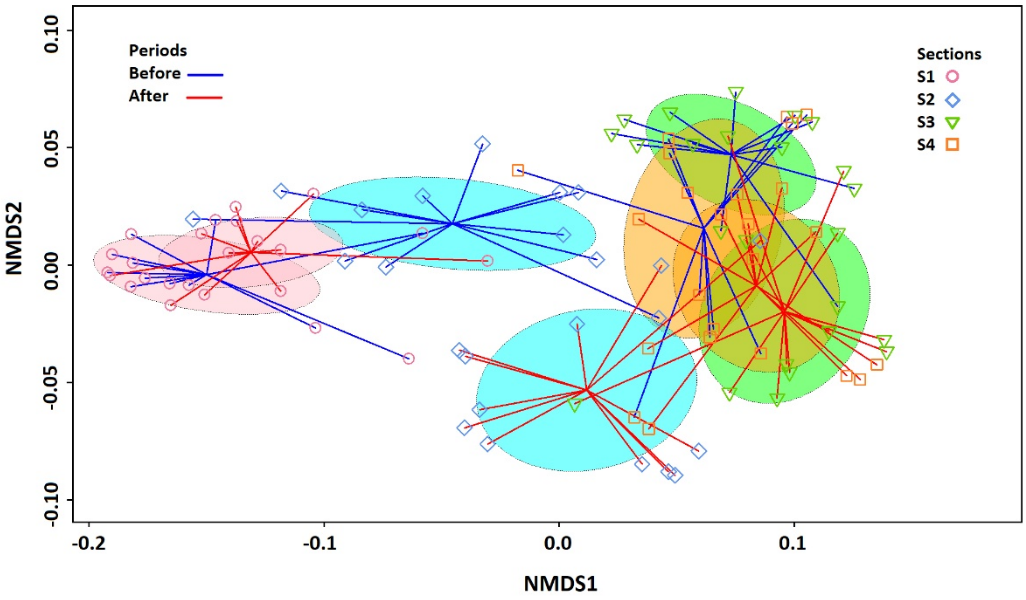


Figure 4.6 NMDS ordination plot (centroids and 95% confidence elipses) of fish communities in Sections 1–4 before (autumn 2017 and spring 2018) and after (spring 2019 and autumn 2019) tidal weir removal. Data for Sections 5 and 6 are not shown because they overlapped greatly with Section 4 and obscured the pattern.

Table 4.6 Pairwise comparisons of fish community based on Bray-Curtis dissimilarity matrix between Sections 1, 2 and 3 during the survey periods, P1 (autumn 2017 and spring 2018), P2 (autumn 2018 and spring 2019) and P3 (spring 2019 and autumn 2019). Significant values in bold.

Period	Sections	Mean square	*df*	*F*	*P*
P1	S1 vs S2	0.968	1,22	13.06	**0.001**
	S1 vs S3	1.204	1,22	19.6	**0.001**
	S2 vs S3	0.383	1,22	4.16	**0.024**
P2	S1 vs S2	0.807	1,22	17.45	**0.001**
	S1 vs S3	1.355	1,22	32.39	**0.001**
	S2 vs S3	0.168	1,22	3.99	**0.024**
P3	S1 vs S2	0.692	1,22	8.72	**0.001**
	S1 vs S3	1.061	1,22	15.53	**0.001**
	S2 vs S3	0.114	1,22	3.58	**0.037**

Table 4.7 PERMANOVA comparisons of fish and invertebrate communities in each section between periods P1 (autumn 2017 and spring 2018, before barrier removal), P2 (autumn 2018 and spring 2019) and P3 (spring 2019 and autumn 2019). Significant values are in bold.

Community	Section	Periods	Mean square	*df*	*F*	*P*
Fish	1	P1 vs P2	0.190	1,22	4.259	**0.020**
		P1 vs P3	0.212	1,22	2.656	**0.032**
	2	P1 vs P2	0.460	1,22	6.077	**0.004**
		P1 vs P3	0.420	1,22	5.683	**0.003**
	3	P1 vs P2	0.195	1,22	3.337	**0.030**
		P1 vs P3	0.255	1,22	5.098	**0.012**
	4	P1 vs P2	0.040	1,22	1.186	0.400
		P1 vs P3	0.061	1,22	1.434	0.264
	5	P1 vs P2	-0.002	1,22	-0.119	1.000
		P1 vs P3	0.239	1,22	5.401	**0.050**
	6	P1 vs P2	0.003	1,22	0.109	0.903
		P1 vs P3	0.059	1,22	1.794	0.253
Invertebrate	1	P1 vs P2	0.537	1,10	2.180	0.027
		P1 vs P3	0.424	1,10	1.850	0.101
	2	P1 vs P2	0.177	1,10	0.821	0.611
		P1 vs P3	0.167	1,10	0.819	0.564
	3	P1 vs P2	0.249	1,10	2.182	0.028
		P1 vs P3	0.168	1,10	1.647	0.107
	4	P1 vs P2	0.102	1,10	0.672	0.737
		P1 vs P3	0.219	1,10	1.631	0.122

4.3.3 Upstream recolonisation by eel and flounder

Although eel were present upstream of the barrier before its removal, they occurred at very low densities (Figure 4.7). Eel densities before barrier removal differed significantly among sampling sections (LMM, $F_{5,60}$ = 29.54, P < 0.001); eel abundance was higher in Section 1 than upstream sections in both seasons (Figure 4.7, Table 4.8; LMM pairwise post hoc, P < 0.001 in all cases). Prior to barrier removal, there was no significant difference in eel density between upstream sampling sections (Section 2–6; LMM pairwise post hoc, P > 0.05 in all cases).

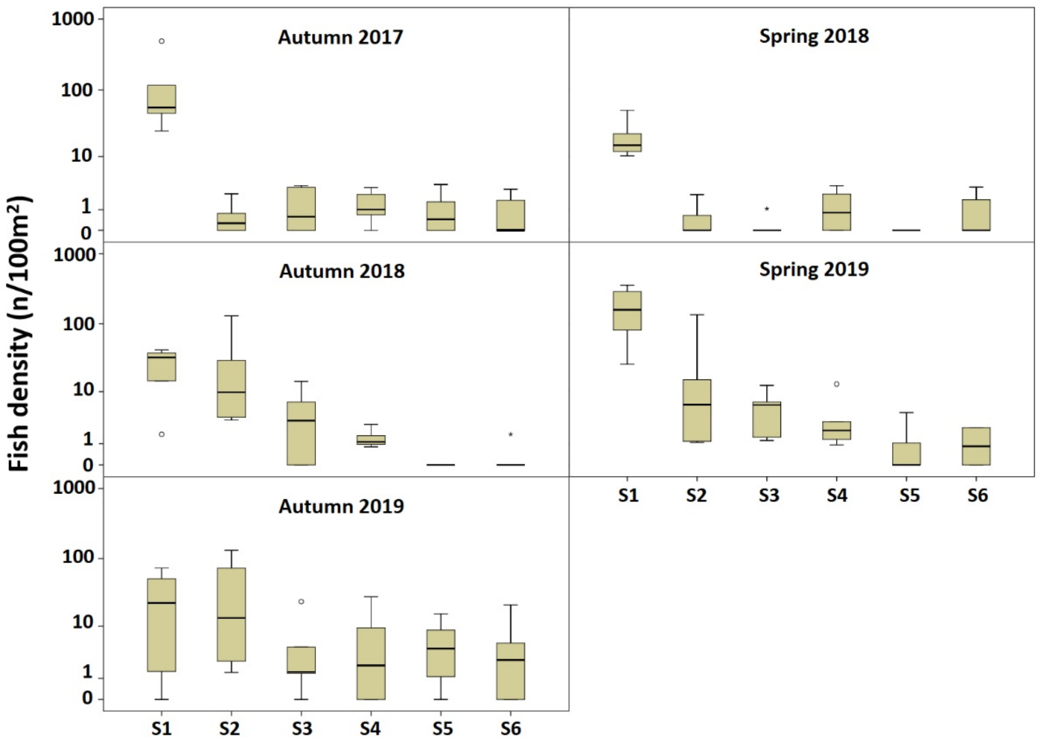


Figure 4.7 Box plots showing median (with quartiles, ranges and outliers) European eel densities (per 100 m^2) in each section before and after tidal barrier removal (removed, after spring 2018 sampling). Section 1 is in the tidal zone, downstream of the barrier, Section 2 is the impounded zone, while Section 6 is the furthest site upstream.

Table 4.8 Pairwise comparisons of eel density between survey sections at different times relative to weir removal. Tukey's multiple comparison test applied. Significant values in bold.

Periods	Before				After					
Season	autumn 2017		spring 2018		autumn 2018		spring 2019		autumn 2019	
Section	*Z*	*P*	*Z*	*P*	*Z*	*P*	*Z*	*P*	*Z*	*P*
2 - 1	-7.06	<**0.001**	-6.11	<**0.001**	-0.44	0.998	-4.62	<**0.001**	0.52	0.995
3 - 1	-6.72	<**0.001**	-6.77	<**0.001**	-3.42	**0.008**	-5.43	<**0.001**	-1.37	0.747
4 - 1	-5.95	<**0.001**	-5.12	<**0.001**	-3.34	**0.011**	-5.93	<**0.001**	-1.54	0.640
5 - 1	-6.86	<**0.001**	-7.36	<**0.001**	-6.63	<**0.001**	-8.27	<**0.001**	-1.15	0.863
6 - 1	-7.35	<**0.001**	-5.90	<**0.001**	-6.04	<**0.001**	-7.70	<**0.001**	-1.61	0.593
3 - 2	0.33	0.999	-0.65	0.987	-2.98	**0.034**	-0.81	0.965	-1.89	0.408
4 - 2	1.11	0.878	0.99	0.920	-2.89	**0.044**	-1.31	0.780	-2.06	0.307
5 - 2	0.20	1.000	-1.25	0.813	-6.19	<**0.001**	-3.65	**0.004**	-1.67	0.552
6 - 2	-0.30	1.000	0.22	1.000	-5.60	<**0.001**	-3.09	**0.025**	-2.13	0.270
4 - 3	0.78	0.972	1.65	0.568	0.09	1.000	-0.50	0.996	-0.17	1.000
5 - 3	-0.13	1.000	-0.59	0.991	-3.21	**0.017**	-2.84	0.052	0.22	1.000
6 - 3	-0.63	0.989	0.87	0.954	-2.62	0.092	-2.27	0.205	-0.24	1.000
5 - 4	-0.91	0.945	-2.24	0.219	-3.29	**0.013**	-2.34	0.177	0.39	0.999
6 - 4	-1.41	0.724	-0.78	0.971	-2.71	0.074	-1.78	0.480	-0.07	1.000
6 - 5	-0.50	0.996	1.46	0.689	0.59	0.992	0.56	0.993	-0.46	0.997

Eel were divided into three length classes: class 1 (40–109 mm; recently recruited glass eel and elvers), class 2 (110–219 mm; those that had spent less than 2 years in freshwater) and class 3 (≥ 220 mm; those that had spent more than 2 years in freshwater) (Domingos *et al.*, 2006). Before weir removal, 47.2% of eel in Section 1 were glass eel / elver, but in the remaining upstream sections only 10.3% of those caught were glass eel / elver (Figure 4.8). Five months after weir removal, mean eel density in Section 2 increased from 0.5 to 32.5 per 100-m^2, significantly higher than sections further upstream

(LMM pairwise post hoc, $P < 0.005$ in all cases, Table 4.8), and 79.2% were glass eel / elver. In Section 2, eel contributed 15.6% of the dissimilarity in fish assemblages in P2 and it increased to 22.6% in P3 (SIMPER). However, eel density in Section 1 remained unchanged following removal of the weir (Table 4.9; LMM, $F_{1,23} = 0.008$, $P = 0.917$).

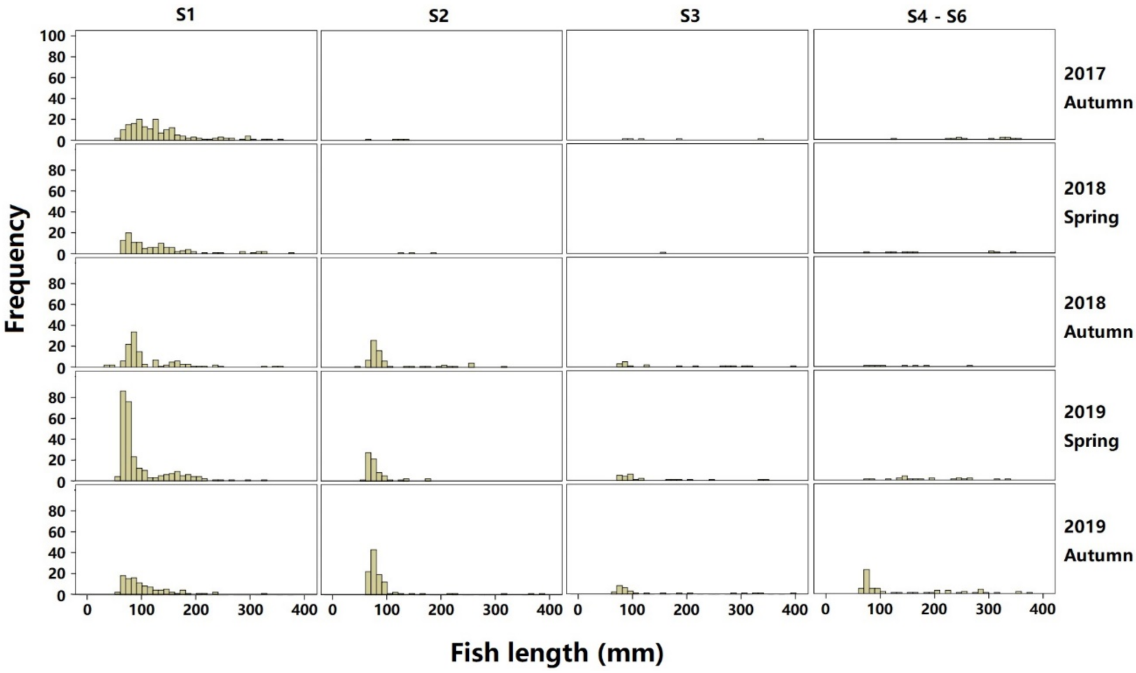


Figure 4.8 Length frequency distribution of European eel in each section before and after weir removal (removed after spring 2018 sampling). Section 1 is the furthest downstream site in the tidal zone, Sections 2–6 are non-tidal, with Section 6 the furthest upstream. Samples in Sections 4-6 have been combined.

Table 4.9 Linear mixed-effects model (LMM) output showing changes in European eel abundance in each stream section before, compared to after, barrier removal (Before: autumn 2017 and spring 2018; After: autumn 2018, spring 2019 and autumn 2019). Season and site were used as random factors in the analysis. Significant values are in bold.

Section	Mean square	*df*	*F*	*P*	*Trend*
All combined	11.874	1,143	24.992	<**0.001**	↑↑↑
1	0.008	1,23	0.011	0.917	**-**
2	16.782	1,23	48.895	<**0.001**	↑↑↑
3	4.90	1,23	14.591	<**0.001**	↑↑↑
4	0.999	1,23	4.673	**0.041**	↑↑
5	0.393	1,22	0.920	0.348	↑
6	0.317	1,23	0.996	0.329	↑

For Sections 4–6, eel abundance did not change markedly until autumn 2019, 17 months after weir removal. In spring 2019, strong eel recruitment was recorded in the tidal zone and 77.9% of the total eel catch comprised the class 1 group. By autumn 2019, it was evident that this year class had colonised the whole stream (Figure 4.8). Even in Section 6, 68.8% of captured eels were class 1. For all upstream sections combined (Sections 2–6), mean eel length in autumn 2019 (114 ± 82 mm) was shorter than in autumn 2017 (247 ± 121 mm; independent *t*-test, $t_{226} = -7.176$, $P < 0.001$). No difference in eel density was found among stream sections in autumn 2019 (LMM, $F_{5,25} = 6.10$, $P = 0.195$). Overall, eel density across all sections increased significantly after barrier removal (Table 4.9; LMM, $F_{1,143} = 11.874$, $P < 0.001$). The fact that eel density did not increase in section1 after barrier removal, reflects that the increase upstream was a result of increased access and redistribution.

Flounder were divided into two length-age classes: 10–80 mm, Age 0-group; 81–140 mm, Age 1-group (Summers, 1979, 1980). Before weir removal, flounder occurred only in the downstream tidal section (Section 1, Figure 4.4), of which 95.7% were 0-group (Figure

4.9). After barrier removal, flounder started colonizing upstream (Figure 4.4, Figure 4.9); mean flounder density at Section 2 increased from zero to 14.9 per 100 m^2 (P1 vs P3). Over the same time period, mean flounder density in Section 1 decreased from 119.9 to 37.6 per 100 m^2 (LMM, $F_{1,27}$ = 4.62, P = 0.04). Flounder were not recorded upstream of Section 2.

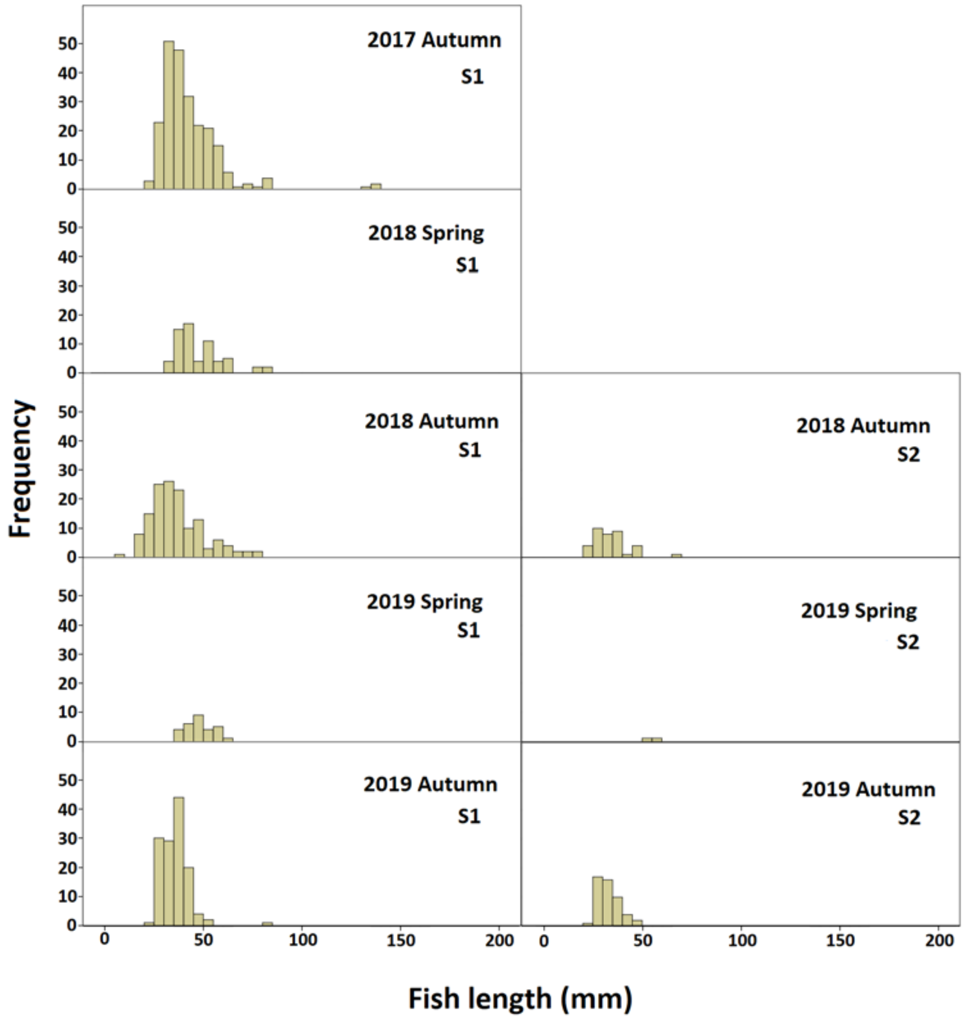


Figure 4.9 Length frequency distribution of flounder in Section 1 (tidal) and Section 2 (impounded) before and after the tidal weir removal (weir was removed after spring 2018 sampling). No flounder were recorded upstream of the weir in autumn 2017 and spring 2018.

4.3.4 Invertebrate community changes post-barrier removal

The benthic invertebrate communities in Sections 1 and 3 differed in P2 compared to the pre-removal period (P1) (Figure 4.10, PERMANOVA, $P < 0.05$ in both cases; Table 4.7), but no differences were evident for any sections between P1 and P3. For Section 1, SIMPER outputs showed that the contribution of three invertebrate taxa (Oligochaeta, Asellidae and Dixidae) changed significantly after weir removal (SIMPER, all $P < 0.05$; Table 4.10). For Section 3, SIMPER revealed that the contribution of three invertebrate families (Baetidae, Hydropsychidae and Heptageniidae) changed significantly post weir removal (SIMPER, all $P < 0.05$, Table 4.10). For the ASPT, no difference was found in each section before and after the weir removal (LMM, $P > 0.05$ in all cases).

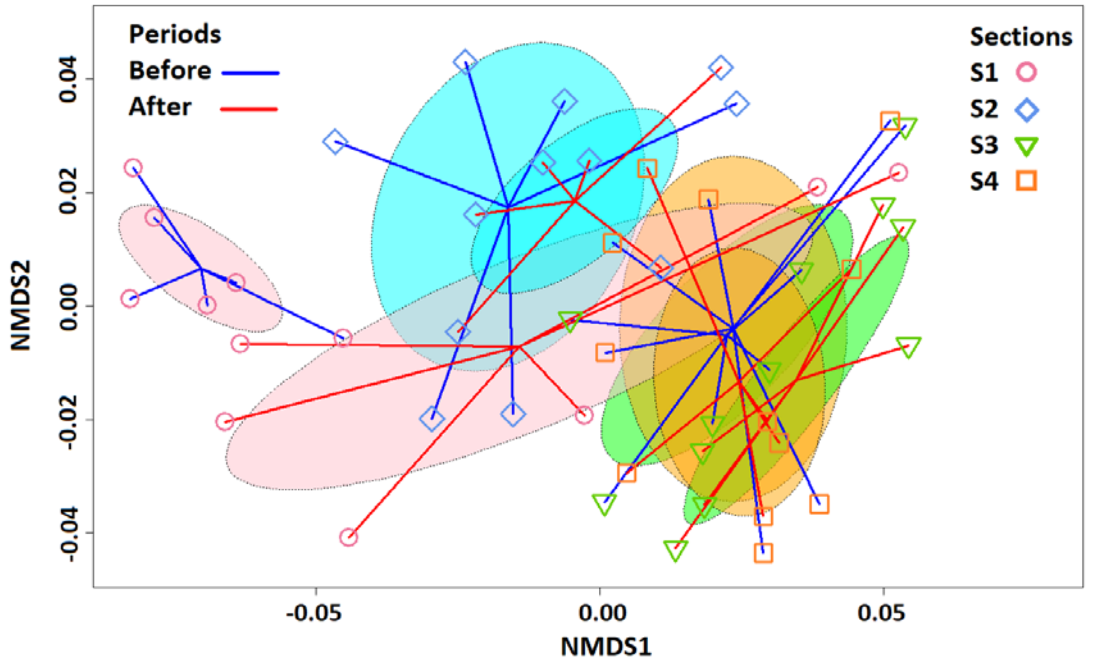


Figure 4.10 NMDS ordination plot (centroids and 95% confidence elipses) of benthic invertebrate communities in Sections 1–4 before (autumn 2017 and spring 2018) and after (autumn 2018 and spring 2019) tidal weir removal.

Table 4.10 Results of SIMPER analyses based on Bray-Curtis dissimilarity index, showing contribution to change of taxa in Sections 1 and 3 between P1 (autumn 2017, spring 2018) and P2 (autumn 2018, spring 2019). Significant values in bold.

Section 1 P1 vs P2	Contribution (%)	*P*	Section 3 P1 vs P2	Contribution (%)	*P*
Oligochaeta	11.90	**0.003**	Baetidae	7.25	**0.005**
Gammaridae	8.40	0.21	Simuliidae	6.15	0.068
Baetidae	7.75	0.064	Hydropsychidae	5.89	**0.039**
Crangonidae	6.45	0.874	Lymnaeidae	5.45	0.729
Chironomidae	6.27	0.299	Chironomidae	4.47	0.092
Lymnaeidae	5.89	0.138	Gammaridae	4.46	0.762
Asellidae	5.80	**0.03**	Oligochaeta	4.15	0.288
Elmidae	5.76	0.751	Tipulidae	4.11	0.992
Glossiphoniidae	3.34	0.064	Heptageniidae	3.86	**0.05**
Arenicolidae	3.28	0.936	Elmidae	3.81	0.721
Tipulidae	3.22	0.231	Glossiphoniidae	3.77	0.687
Mysidacea	2.97	0.976	Planorbidae	3.75	0.691
Hydropsychidae	2.73	0.078	Leptoceridae	3.47	0.676
Gyrinidae	2.32	0.559	Beraeidae	3.29	0.103
Perlodidae	2.16	0.067	Perlodidae	3.23	0.168
Heptageniidae	1.89	0.078	Asellidae	3.19	0.371
Simuliidae	1.89	0.078	Goeridae	3.10	0.068
Planorbidae	1.81	0.47	Muscidae	3.06	0.073
Dixidae	1.80	**0.011**	Sphaeriidae	3.00	0.401
Limnephilidae	1.74	0.078	Dytiscidae	2.89	0.9
Empididae	1.73	0.078	Physidae	2.72	0.168
Glossosomatidae	1.45	0.081	Limnephilidae	2.63	0.274
Goeridae	1.32	0.923	Ancylidae	2.12	0.429
Sphaeriidae	1.32	0.082	Ephemeridae	2.00	0.474
Corophiidae	1.19	0.965	Glossosomatidae	1.74	0.141
Dytiscidae	1.08	0.981	Sialidae	1.53	0.894
Sphaeromatidae	0.99	0.965	Lepidostomatidae	1.01	0.403
Ancylidae.	0.91	0.981	Sericostomatidae	0.86	0.184
Muscidae	0.91	0.981	Gyrinidae	0.82	0.263
Leptoceridae	0.89	0.081	Valvatidae	0.81	0.745
Phryganeidae	0.84	0.082	Phryganeidae	0.80	0.403
			Ephemerellidae	0.65	0.941

4.4 Discussion

This is the first study to report on the effects of removal of a small tidal-limit barrier on adjacent aquatic habitat, and on the fish and invertebrate communities. Results of this study show that although the small tidal barrier did not fully prevent eel passage, it dramatically reduced upstream European eel abundance and altered eel size structure within the upstream reach. The study provides evidence that removal of the barrier reinstated longitudinal connectivity effectively, and without unforeseen consequences. Increased habitat diversity was created immediately upstream and, although large amounts of silt were mobilised, most was transported through the system within a year. Effects on the benthic invertebrate community appear to have been minor and transient. Despite the short study period and the lack of a nearby dammed control stream (no similar study system existed which could have provided a comparator; tidal weir vs no tidal weir), strong benefits of the barrier removal were evident for the fish community, in terms of their ability to redistribute and colonise suitable habitat, potentially in areas with low competition. The density of European eel, particularly new recruits (glass eel and eel elver), increased in all five upstream sections, and the total fish density in the previously impounded zone also increased after barrier removal. Pre- barrier removal, three-spined stickleback dominated the impounded zone. This species is typical of slow-moving water and is often dominant in degraded, ponded habitat (Wootton *et al.*, 1978). Following the weir removal, resident fishes such as bullhead and stone loach benefitted from the lotic habitat (Tomlinson and Perrow, 2003; Freyhof, 2013), and their abundance increased. Bullhead (*Cottus gobio* species complex) is protected under the European Species and Habitats Directive Annex II and is a Biodiversity Action Plan species in the UK, so the return of lotic conditions by barrier removal can provide a tool to support this species' recovery in degraded lowland streams, especially since it has poor dispersal abilities (Tummers *et al.*, 2016).

This study suggests that (1) the removal of the tidal-limit barrier restored more suitable habitat for migratory and resident fishes, (2) free passage to upstream nursery habitat was restored, and (3) post-weir removal, the upstream recolonization and recruitment of eel was greatly increased within two years. Evidence is growing rapidly that stream barrier removal can be very effective for aquatic conservation (Catalano *et al.*, 2007; Burroughs *et al.*, 2010; Birnie-Gauvin *et al.*, 2018; Ding *et al.*, 2018). Where possible, barrier removal should be one of the first tools in the conservationist's 'toolbox' to be used for stream connectivity restoration (Garcia De Leaniz, 2008; Tummers *et al.*, 2016; Birnie-Gauvin *et al.*, 2017c). Earlier debate over the tradeoff of risks and benefits of barrier removal concentrated particularly upon medium- and large-sized dams, where the removal costs are relatively high and especially centred upon the risks of contaminated and uncontaminated fine sediment release from the impoundment (Bednarek, 2001; Poff and Hart, 2002). That risk applies much less to small barriers, which do not retain large amounts of fine-sediment deposits behind them. Yet the vast majority of artificial river barriers are small (Januchowski-Hartley *et al.*, 2013; Jones *et al.*, 2019; Sun *et al.*, 2020). Nevertheless, although removal of redundant barriers is a preferred restoration tool, in many cases barriers cannot be removed due to societal needs or because of constraints such as erosion risks on nearby infrastructure (Birnie-Gauvin *et al.*, 2017c). On the River Tees, only two out of twenty barriers where connectivity restoration has occurred have been removed, the remainder were installed with fish passes (Sun *et al.*, 2020). Yet such a proportion is probably typical of European and North American rivers. As evidence of the benefit to cost ratio of stream barrier removal increases and confidence grows, hopefully efforts will increasingly be concentrated on achieving barrier removal.

4.4.1 Response of the fish and invertebrate communities

The single-most important indicator of the success of tidal barrier removal in this study was the rapid recolonisation of most of the stream by juvenile eel, suggesting it can have similar benefits elsewhere. Tamario *et al*. (2019) provided evidence that fishway types, other than nature-like bypasses, have no better effect on eel distribution upstream of dams than dams with no fishways. Their study was unable to evaluate the benefits of barrier removal due to small sample size. We recommend that eel conservation measures are likely to benefit disproportionately from investment in removal of redundant barriers and provision of nature-like bypasses in the lower reaches of rivers. The importance of unimpeded passage of diadromous fishes, especially in the lower reaches of catchments, is widely acknowledged (Kemp and O'Hanley, 2010; Nunn and Cowx, 2012).

Tidal-limit barrier removal allowed for rapid upstream immigration of juvenile eel from the tidal reach. Although a 1-m barrier, approximately 1.5 km upstream, remained, it is evident that its size, form and possibly the addition of a pool fishway and bristle-type eel pass, did not impede passage of eel smaller than 110 mm past it. After the previously impounded reach was restored to shallower lotic habitat, it may also have become more suitable for eel to colonise. Recent research has shown that European eel in lotic waters prefer to use shallow and rocky habitat such as riffle and run rather than deep pool habitat (Acou *et al.*, 2011). Use of shallow habitat can also potentially reduce the chance of small eel being predated (Degerman *et al.*, 2019). Mean eel length in autumn 2019 was shorter than in autumn 2017, suggesting that young recruits (glass eel and elver) were primarily responsible for the increase in the upstream eel population. In Section 6, the furthest upstream site, more than 60% of captured eel were under 110 mm in length in autumn 2019. A dam removal study in America found that dam removal significantly increased American eel (*Anguilla rostrata*) abundance in headwater streams, and immigration of small individuals (< 300 mm) was primarily responsible for the observed increases in eel numbers (Hitt *et al.*, 2012).

The study stream, Claxton Beck, flows into the Tees estuary downstream of the Tees Barrage, which opened in 1995, and was built as part of an urban economic redevelopment plan. That tidal barrage has a salmon ladder, navigation lock and a bristle pass for eels but represents a major barrier for upstream eel migration to most of the Tees catchment. The rapid increase in eel density and distribution through Claxton Beck shows how such restorative actions can contribute towards eel management plans for individual catchments such as the Tees, part of the Northumbria RBD.

In autumn 2018, after the weir removal, although there was no significant difference in total fish density in Section 1, downstream of the barrier’s former position, compared to before removal, a decrease in flounder abundance occurred. This is likely because a large amount of silt was released to the downstream section after barrier removal and covered the previously suitable sandy habitat. Juvenile flounder have been observed prefer to use sandy and gravelly substrate (Le Pichon *et al.*, 2014), so it is likely that after weir removal some flounder in Section 1, especially in those patches most affected, moved upstream or further downstream to more suitable habitat. Indeed, flounder rapidly colonized the formerly impounded reach soon after barrier removal. On the other hand, the downstream invertebrate community only showed differences in the first period, and the invertebrate community changes in Section 3 may have been caused by moving the sampling location post-weir removal. Any change in invertebrate communities seems to have been transient, perhaps due to initial sediment mobilisation soon after weir removal. This suggests that downstream river habitat recovered within 17 months. Also there was no significant change in the ASPT, suggesting weir removal did not degrade the downstream water quality. In contrast to flounder, the eel population in the tidal reach was not affected by the temporary increase in fine sediment. This is likely because eel is more tolerant to muddy substrate and elver often use soft substrates as shelter in which to hide (J. Sun, pers. obs.). In addition, eels may hibernate in soft muddy substrate when the water temperatures drops to less than 8-9 °C (Degerman *et al.*, 2019).

In contrast to eel, and to the study of Birnie-Gauvin *et al.* (2018), the population of brown trout in Claxton Beck has not yet benefitted from barrier removal. Brown trout is a species of 'principal importance' for biodiversity conservation in England and Wales under Section 41 of the Natural Environment and Communities Act 2006. In 1997, the Environment Agency stocked approximately 10,000 brown trout fry upstream of Section 6, but fish surveys close to the release site in 1998, 2000 and 2004, caught only small numbers of trout (R. Jenkins, unpublished data). Apart from trout, low abundance of bullhead, three-spined stickleback and stone loach were caught at Section 6 during previous fish surveys (see section 2.3.3.6). During this study, a few juvenile and adult brown trout were caught before weir removal. No significant changes were found in the trout population in the post-weir-removal surveys, and no Age 0 trout were caught in 2019. Although adult sea trout can easily immigrate from the Tees estuary it is possible that few were doing so during the study. The philopatric nature of sea trout (Lucas and Baras, 2001) would also tend to result in slow recolonisation if the existing population is small. It is also the case that although the previously impounded reach became shallower and more diverse in habitat types, the bed was mostly of sand and compacted earth, which is unsuitable spawning and suboptimal juvenile habitat for trout (Louhi *et al.*, 2008). In the upper reach (Section 6), although riffles with gravel occurred and lotic specialists such as bullhead were common, it is possible that interstitial fine sediment might be too abundant, and interstitial oxygen supply too poor, to enable trout egg survival and development (Kemp *et al.*, 2011). Indeed, three-spined stickleback, a species typically associated with degraded water quality and habitat were also abundant at this site. Enhanced connectivity, without sufficient improvement in habitat quality and water quality cannot achieve desired restoration outcomes (Roni *et al.*, 2008; Tummers *et al.*, 2016) and needs to be a focus in this intensively farmed subcatchment in the future. However, it is also possible that unlike the observations of Birnie-Gauvin *et al*. (2018), recovery of trout populations in Claxton Beck will take much longer than the short duration of this study. This reflects the importance of standardised long-term monitoring for charting recovery of fish populations (see Chapter 2), particularly due to inherent stochasticity in such processes. It is anticipated, or hoped, that the Environment Agency or Tees Rivers Trust will continue longer-term standardised quantitative sampling of the fish communities in order to take advantage of the robust but short-term dataset provided here.

4.5 Conclusions

This study suggests that, for small tidal and tidal-limit barriers in temperate climates, barrier removal is an appropriate method by which to restore aquatic habitat and increase the abundance of both resident and migratory fish species, especially benefitting eels. Findings of this study support the recent emphasis on barrier removal as a very powerful tool for river restoration, and have important implications for environmental agencies engaged in river and estuary management. In addition, this study also showed that barrier removal can be an effective method in the management of priority conservation species like globally threatened European eel. The apparent success of barrier removal for reconnecting habitats for European eel, albeit at the small scale of this study, and for American eel (Hitt *et al.*, 2012), suggests that it should be trialled for other eel species.

Chapter Five

Does connectivity restoration restore natural fish communities in degraded subcatchments of post-industrial rivers?

This chapter contains a modified and extended version of a manuscript planned for submission to Science of the Total Environment: Sun, J., Tummers, J., Galib, S.M. & Lucas, M.C. Does connectivity restoration restore natural fish communities in degraded subcatchments of post-industrial rivers?

Statement

J. Sun and M. Lucas conceived the study and experimental design, J. Sun led the fieldwork except for 2012-2016 raw data that were provided by J. Tummers and M. Lucas. J. Sun analysed the data and wrote the chapter, with comments provided by M. Lucas. Supporting historical fish raw data were provided by the Environment Agency.

Summary

River barrier removal and fish pass construction have been increasingly used as management tools to restore river connectivity, but few studies have documented changes in fish abundance following catchment/subcatchment scale connectivity restoration, and the efficacy of connectivity restoration on fish communities is not fully understood. In this study, several types of before-after experimental design methodology (e.g. before-after-downstream-upstream), were used to determine the effects of multiple barrier removal and fish pass installation on fish communities in three heavily degraded streams in the River Wear, Northeast England. Multiple sites on streams were sampled by electric fishing once a year, in summer, from 2013 to 2019 (River Deerness) and over shorter periods for Brancepeth Beck, Cong Burn and Bedburn Beck.

Different outcomes were observed between the three streams undergoing restoration. In the River Deerness, at sites where connectivity was restored (especially where barrier removal occurred) the fish community benefitted. Total fish abundance significantly increased three years after the restoration and remained elevated to the end of the study in 2019. Both brown trout (*Salmo trutta)* and bullhead (*Cottus perifretum)* benefitted from the restoration, mean brown trout density increased from 20.9 ± 6.3 per $100m^2$ to 33.8 ± 16.8 per $100m^2$ from 2013 to 2019, largely due to increased Young-of-Year (YoY) trout density, which increased from 10.6 ± 4.6 per $100m^2$ to 19.8 ± 11.8 per $100m^2$. Density of bullhead, a poorly-dispersing species, increased from 4.6 ± 2.7 per $100m^2$ to 32.6 ± 17.9 per $100m^2$ from 2013 to 2019. However, no Atlantic salmon (*Salmo salar*) were recorded over the study timescale. These changes were found to be linked to increases in reproductive success, itself linked to changes in habitat, as well as increased dispersal and migratory capacity.

In Brancepeth Beck, significant differences in the fish community were found after the connectivity restoration, where a portion of barriers in the lower part of the beck were modified or removed. Brown trout, bullhead and stone loach (*Barbatula barbatula)* benefitted from the restoration. Fish density in the restored reach was significantly higher compared with an unrestored reach further upstream (mean trout density in the restored

reach in 2019: 93.7 ± 74.5 per 100m^2; in the unrestored reach: 3.7 ± 4.9 per 100m^2). Within the unrestored reach, the trout density gradually decreased during the study period, from 22.4 ± 10.3 per 100m^2 in 2014 to 3.7 ± 4.9 per 100m^2 in 2019.

In Cong Burn, brown trout density increased in the first few years after the connectivity restoration, then significantly reduced during the study period. Apart from the most downstream sites, fish species richness was extremely low in this stream. This suggests that colonisation access and/or suitable habitat conditions may be available for trout and eel (*Anguilla anguilla*), but not for other resident species, and habitat conditions may not always be suitable for trout migration and reproduction, indicating further habitat and connectivity management is needed.

In Bedburn Beck, a reference stream for this study, both Atlantic salmon and brown trout abundance showed a decreasing trend over recent years. Autumn flows in the middle and lower Wear streams were all relatively low in 2016, 2017 and 2018, potentially impacting access of adult salmonids to the study streams to spawn. Trout recruitment in Cong Burn and, to a lesser degree, Brancepeth Beck, was impacted, while the Deerness remained relatively unaffected, possibly reflecting the early stage of trout population recovery in Cong Burn especially. In Bedburn Beck, in the middle reaches of the Wear, autumn flows were also low in this stream beween 2016 and 2018, potentially impacting access of adult salmonids. Atlantic salmon were found in this stream (second or third most abundant fish at most sites) but were rare or absent in the connectivity-restored streams, possibly reflecting habitat limitations for recovery or a much slower timescale of recovery for salmon, potentially including hysteresis effects.

The results suggest that, in rivers with good aquatic habitat, including good water quality, restoring river connectivity, can be beneficial for both resident and migratory fishes. Compared with fish pass installation, barrier removal is more effective in restoring fish communities in the immediately upstream reach. In some cases, wider catchment management is required along with connectivity restoration to gain a better outcome for fish species sensitive to degraded habitat and/or water quality such as Atlantic salmon and brown trout. The benefits of partial connectivity restoration (i.e. complete removal of a proportion of barriers, provision of fish passage at some or all barriers, or a combination of these) in some streams, especially in those with many barriers, may take many years to develop, especially for species present only in the lower stream and with low dispersal ability to recolonize upstream, such as bullhead.

5.1 Introduction

Globally, rivers are affected by a wide range of pressures including pollution, water abstraction, invasive species and physical modification (Gregory, 2006). In a degraded river system, many types of problems may exist (e.g. water quality, habitat, fine sediment, barriers), and their cumulative effects will increase the degree of habitat degradation (Karr *et al.*, 1985; Casatti *et al.*, 2006; Hermoso *et al.*, 2011). River restoration is defined as the process to return a river section to a near-natural status (Woolsey *et al.*, 2007). This approach has become a priority for river management in many countries such as: Switzerland (Woolsey *et al.*, 2007), Japan (Nakamura *et al.*, 2006), the US (Bernhardt *et al.*, 2005) and the UK (Smith *et al.*, 2014). Identifying the types and impacts of threats in a river is important for developing an effective restoration strategy (Chapter 1). Strategies should be adopted that assess and solve the range of pressures in the river system to enable a full recovery. For example, even after habitat heterogeneity has been increased in a structurally degraded river, some pressures including water pollution and inadequate sediment quality may persist and limit the development of the aquatic community towards its natural state (Lake *et al.*, 2007; Sundermann *et al.*, 2011). In such cases, degraded water quality and sediment issues need to be controlled together with physical restoration to ensure recovery.

In the UK, rivers and streams have been subject to multiple threats over the past two centuries and often longer (Chapter 2). In recent years, water pollution in many UK rivers, especially from point sources, has been reduced due to a variety of regulations (see Chapter 2 for more details). Following improved water quality, a key next step to improve natural fish biodiversity in industrially degraded rivers, is to restore longitudinal connectivity, helping to reinstate migration routes for threatened fish species and restore natural hydromorphic and ecological processes (Addy *et al.*, 2016). In many cases, restoration of key habitat types such as gravel riffles for spawning, or deep pools with natural cover (e.g. boulders, tree roots and canopy, large woody debris) for refuge may also be required to enable recovery of fish populations (Harper *et al.*, 1998; Solazzi *et al.*, 2000; Speed *et al.*, 2016).

In-stream barriers have several negative impacts on fish communities (see Chapters 3 and 4 for more details). They can prevent or delay fish migration and dispersal; alter upstream habitat, causing it to become more lentic; inhibit sediment transport; and indirectly alter biotic elements of habitat upon which fish depend (Mueller *et al.*, 2011). In low-gradient streams, such as in Denmark, habitat altering effects may be important (Birnie-Gauvin *et al.*, 2017a), but in upland streams such as those of the Pennine rivers in this thesis, barrier effects may be more important. The height of a barrier is not the only factor that will affect fish from ascending to upstream habitat, the water depth immediately below the structure and the water depth on the barrier surface also play an important role in fish passage. In some cases, adult salmonids were unable to pass a 45 cm high structure due to the insufficient water depth below the obstacle and on the obstacle itself (Ovidio and Philippart, 2002). For example, brown trout require the water depth immediately below the barrier to be at least twice the maximum body depth of the fish, to enable the fish to gain momentum and leap (Ovidio and Philippart, 2002; Baudoin *et al.*, 2014). More generally, minimum water depth for barrier passage has been set at 100-150% of fish body depth, dependent on barrier type (Ovidio and Philippart, 2002; Baudoin *et al*., 2014).

A single barrier may cause some delay or prevent the upstream movement of migratory fish. If there are multiple barriers present in a single river, the accumulative effects on fish movement could be even stronger. A study on river lamprey (*Lampetra fluviatilis*) in North East England found that only 1.8% of spawners managed to ascend five successive low-head barriers to reach the zone where 98% of spawning habitat occurred (Lucas *et al.*, 2009). Another study, on sockeye salmon (*Oncorhynchus nerka*), has also shown the cumulative effects of fish passage at multiple barriers could significantly reduce migration speed (Naughton *et al.*, 2005). Even if 90% of fish ascend each of a series of barriers, the proportion above the most upstream is just 59% of the starting cohort, yet if most of a critical habitat is upstream of that, the population will be compromised. The ability to pass multiple barriers also has major consequences for fish dispersal, including the recolonization of rivers and streams from which native fish populations have been extirpated, especially when recolonization occurs in an upstream direction by population fragments (Radinger and Wolter, 2014; Tummers *et al.*, 2016; Wilkes *et al.*, 2019).

When fish attempt to ascend a barrier, the increased locomotory effort results in increased energy expenditure (Newton *et al.*, 2018). The energy lost in fish associated with obstacle passage may lead to a subsequent cost on gonad production as well as the spawning activity and eventually reduce the evolutionary fitness of the fish (Newton *et al.*, 2018). For species such as Atlantic salmon and river lamprey, adult fish stop feeding at river entry, so they would not be able to recover any lost energy during the migration period (Lucas and Baras, 2001). Normally, the migration and spawning process would lead to an approximate 40% loss in body weight of Atlantic salmon (Hendry and Cragg-Hine, 2003). If the process was delayed due to obstruction, then adult fish may spend more energy and lose more body weight before they reach the spawning ground, causing increased mortality prior to spawning. Obstacles to movement often also cause aggregation of fish approaching the barrier and this can result in locally high predator densities, difficult in avoiding predators, and high rates of predation, further impacting on fish populations (Newton *et al.*, 2019).

Considerable efforts have been made to restore river connectivity globally, and several methods have been developed to improve fish passage (see Chapter 1 for full discussion). Among all the methods, barrier removal has become an increasingly important approach in the river restoration. Total removal of a barrier (see Chapter 4 for fuller discussion) is considered to be the only feasible method in restoring both fish passage and river habitat (Birnie-Gauvin *et al.*, 2017a; Dodd *et al.*, 2017). Barrier removal returns the flow conditions in a previously impounded reach from lentic to lotic and restores sediment transport to the formerly impounded reach, recreating riffle/pool habitat (if the gradient is sufficient), exposing larger substrates such as pebbles and cobbles, and increases biodiversity (Bednarek, 2001, see also Chapter 4). Several recent studies have shown the benefits of barrier removal on restoring stream fish populations (Burroughs *et al.*, 2010; Fjeldstad *et al.*, 2012; Birnie-Gauvin *et al.*, 2017b, 2018).

When multiple barriers are present in a single river, understanding how passage improvements at different obstacles, interact to affect river connectivity is important to sustainable river management and conservation of migratory fish species (Fullerton *et al.*, 2010; King and O'Hanley, 2016). However, very few studies have been published on multiple barrier removals over the whole catchment / subcatchment. Birnie-Gauvin *et al.* (2018) found that removing six weirs in a Danish river (seven weirs present in total, the upstream most remained) increased numbers of adult brown trout spawners and smolt abundance in the downstream reaches, along with a decreased average fish length (indicative of increased production of young and/or emigration of larger, older fish) and indications of an earlier peak downstream migration of smolts. However, they did not report any changes of fish abundance in the unrestored upstream reach during the study period. Tummers *et al.* (2016) found slight increases in age 0+trout and total bullhead densities, and slight decrease in stone loach densities immediately upstream of connectivity-restored structures; and fish assemblage remained similar at five of six connectivity-restored sites in the River Deerness, NE England (nine barriers present in total, seven barriers were removed/modified) in a short-term (2–3 years) study. A dam removal study in the Sedgeunkedunk Stream in the US (three dams, one removed, one replaced with rockramp fish pass, one remained) showed significant decreases in fish species richness and abundance downstream of the former dam site and a corresponding increase in fish abundance upstream of the former dam site (Gardner *et al.*, 2013). In addition, fish density and biomass at the unrestored site remained low.

Although barrier removal is the desired approach in restoring longitudinal connectivity, it is not always feasible due to reasons such as financial costs, flood control, hydropower, water supply, irrigation, and recreation (Kuby *et al.*, 2005). An alternative plan to improve river connectivity and reduce the negative effects of a barrier on fish migration is fish pass or easement installation. These are mitigations rather than full solutions. In recent years, lots of effort has been made in developing and constructing fish passes to mitigate the effects of these barriers to fish movement (Silva *et al.*, 2018). In the northern hemisphere, the majority of fish pass design is still focused on facilitating the migration of anadromous salmonid fishes and clupeids (Dodd *et al.*, 2017; Silva *et al*., 2018). But according to recent studies, the efficiency of some fish passes in facilitating fish migration remains low

(Bunt *et al.*, 2012; Noonan *et al.*, 2012), even for salmonids in some cases (Lothian *et al.*, 2020). Also many of the studies focus on the passage efficiency for particular fish species rather than the changes in fish communities after fish pass installation. In recent years it has been argued that, to be properly effective in ecological restoration, fish passes (and easements) need to facilitate dispersal of weakly swimming fish species in the community, as well as more strongly swimming migratory species such as salmonids (Tummers *et al.*, 2016; Silva *et al.*, 2018). Such fish passes include nature-like bypasses, with their wide variety of hydraulic conditions and natural habitat features (Bunt *et al.*, 2012; Katopodis and Williams, 2012; Bretón *et al.*, 2013; Baki *et al.*, 2016).

When assessing the success of connectivity restoration, many studies have focused on the restoration effects on migratory species only but ignored resident fish species (Birnie-Gauvin *et al.*, 2017b, 2020; Dodd *et al.*, 2017). Another issue is that most stream restoration projects are small-scale, limited to localised interventions in stream channels and rarely involve extensive interventions at the catchment scale (Lake *et al.*, 2007). The same is often true of quantitative monitoring and experimental evaluation of the efficacy of river reach or catchment restoration. For example, some of these studies only focused on the change near the restored area (Birnie-Gauvin *et al.*, 2017b; Ding *et al.*, 2018) rather than the whole catchment or subcatchment. This reflects that large-scale catchment-scale restoration projects are still mostly lacking (Lake *et al.*, 2007); due to the budget costs they are normally carried out on a piecemeal basis. Catchment-scale connectivity restoration treatment and control comparisons (including Before After Control Intervention 'BACI' comparisons) are very rare (Lake *et al.*, 2007). Normally river restoration needs to have its effect across whole subcatchments, or even catchments, to achieve the full hydromorphic, water quality and ecological benefits.

Because responses to catchment or subcatchment restoration (even for small streams) can take many years, this does not lend itself to the timescale of operation of most research projects (including PhD studies) that are normally for 1-4 years. Also, when assessing the restoration success in a river system, a critical concern is that fish populations are subject to stochastic events. For example, if post-restoration monitoring is conducted in a couple of “good years” (e.g. high flows during spawning migration, facilitating access to spawning sites, but normal flows during critical early life stages of river fish), the positive result can say “restoration worked”, but if monitoring was in a couple of bad years then the outcome could appear to be “restoration failed”. So, evaluation timescale ideally needs to integrate over an extended timescale, allowing for stochasticity. This is problematic, as most studies do not have the capacity to do this over long periods. So too is the ability to properly control for restoration interventions at the catchment scale, by comparison to degraded catchments without restoration, and unimpacted natural reference catchments. Catchments, or even reaches, differ inherently from one another so equal comparisons are difficult, while replication is also problematic, particularly in small-budget studies such as this doctoral project.

In this chapter, the overall aim is to evaluate the effects of subcatchment-scale connectivity restoration on native fish communities in degraded post-industrial rivers. The constituent aims were to determine: (1) in small post-industrial rivers and streams, to what degree does restoration require concerted changes over the whole subcatchment; (2) to what extent can single-site mitigations have beneficial effects on native fish diversity and abundance; (3) how well, in terms of diversity and abundance, and how quickly can the fish community respond to connectivity restoration; and (4) to what degree may barrier removal give better fish community restoration outcomes than fish pass / easement installation, in the reach immediately upstream of the barrier location.

The River Wear catchment was selected for this study. The Wear is one of the most important Atlantic salmon and sea trout rivers in England (see Chapter 2), but many tributaries remain degraded due to pollution (Chapter 2), barriers (Chapter 3) and habitat modification, resulting from industrial, urban and agricultural development, especially within the middle and lower reaches of the river. A significant issue concerning ecological restoration of the catchment is the number of barriers to fish migration and dispersal, particularly within tributaries, which are important nursery areas for salmonids. It is estimated that there are nearly 500 instream artificial barriers still present in the catchment (Chapter 3). These barriers degrade river habitat and cause significant negative impacts on the local fish communities (Chapter 2). Four low-altitude sub-catchments (Cong Burn, River Deerness, Brancepeth Beck and Bedburn Beck) were chosen from the Wear Catchment (Figure 5.1). The first three streams were badly affected by artificial barriers and multiple connectivity restoration projects were conducted in these rivers in an attempt to restore the aquatic connectivity and fish populations. These streams also vary in terms of their historic and current water quality, the Cong Burn in particular having suffered poor water quality in the recent past. By contrast, Bedburn Beck was selected as a relatively natural reference river ('control'). Although it does have a few barriers, it is less affected by artificial barriers than the others, it has had relatively light development and has a long history of good water quality. Although further upstream than the other subcatchments, Bedburn Beck is also relatively low-altitude. It has relatively natural instream habitat and its river substrates provide high quality spawning and rearing habitat for fish like Atlantic salmon and sea trout.

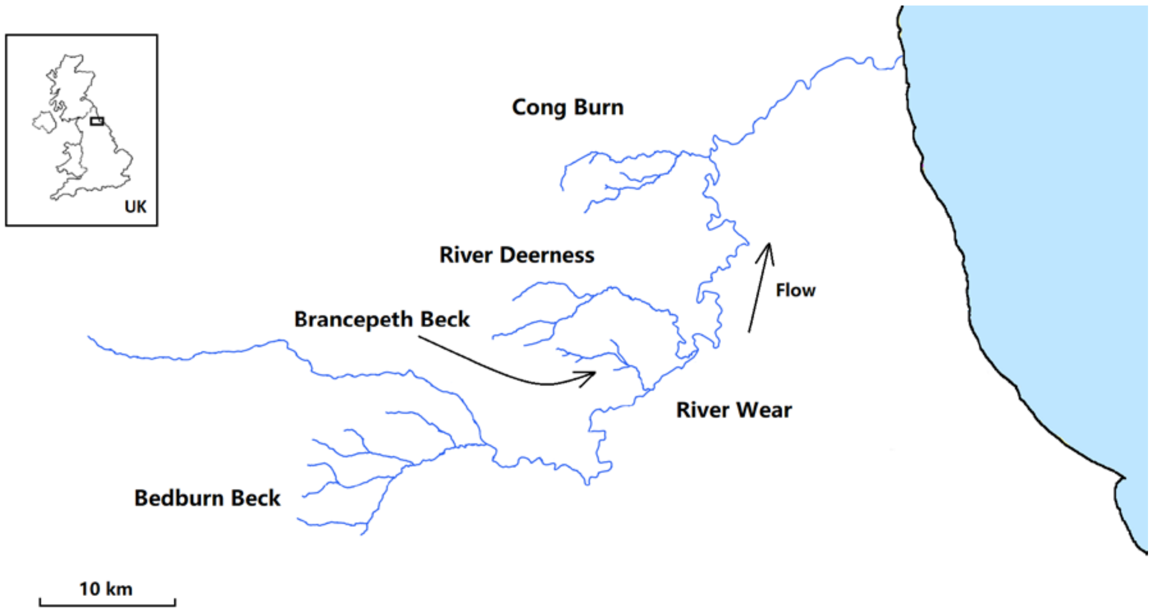


Figure 5.1 Wear catchment and the location of study reaches. For clarity and simplicity the figure deliberately omits other tributaries along this part of the Wear, but these can be seen in Figure 3.1, section 3.2.1.

5.2 Study sites

5.2.1 River Deerness

The River Deerness, located in northeast England, has a source 285 m above sea level, and a mean gradient of 12 m per km, over the 18.74 km that it flows eastwards before joining the lower River Browney at Langley Moor (Figure 5.2). The Deerness subcatchment covers an area of 52.89 km^2, with land use mostly consisting of semi-natural woodland, improved grassland and agricultural land. The stream is characterized by pool-riffle sequences with large areas of gravel, cobble and some boulders, providing good salmonid spawning and nursery habitat, although many areas contain industrial rubble material, including broken tiles and bricks. There are several villages along its course, which originated as coal-mining “pit” villages. The river suffered severe pollution due to coal mining, coal washing and cokeworks, as well as human sewage, from the middle of the 19th Century. Several mines opened in the Deerness area in the mid-19th Century. Cornsay Colliery mine, adjacent to Hedleyhope Beck, a Deerness tributary, opened in 1868. Esh Winning mine, adjacent to Priest Burn, a Deerness tributary, opened in 1866. Waterhouses Colliery opened in 1859, Ushaw Moor mine opened in 1865 and New Brancepeth Colliery opened in 1858, also in the Deerness catchment (Durham Mining Museum, 2020). At the end of the 19th Century, Esh Winning and Waterhouses colliery were producing 100,000 -120,000 tons of coke annually (Emery, 1984). Cornsay Colliery and New Brancepeth Colliery closed in 1953, Ushaw Moor colliery closed in 1960, Waterhouses colliery closed in 1966 and Esh Winning colliery closed in 1968 (Emery, 1984; Durham Mining Museum, 2020). After these collieries closed, polluting material from spoil heaps still leached into the Deerness, causing continued pollution (Brown, 1974).

The water quality in the Deerness did not improve markedly until these coal mines closed and remediation actions started in the early 1970s. Changes in Deerness water quality and fish community are presented in sections 2.3.2.4 and 2.3.2.5, based on historical data. The overall ecological status of the River Deerness was classified as “Poor” by the Environment Agency between 2013 and 2016 (Environment Agency, 2020a). This was partly due to the poor status of the fish community, deficient in salmonid numbers, and largely attributed to poor fish passage in the upper catchment. It was also associated with poor status of macrophytes and phytobenthos in the lower catchment and high phosphate in the sewage discharge downstream of Ushaw Moor bridge (Environment Agency, 2020a).

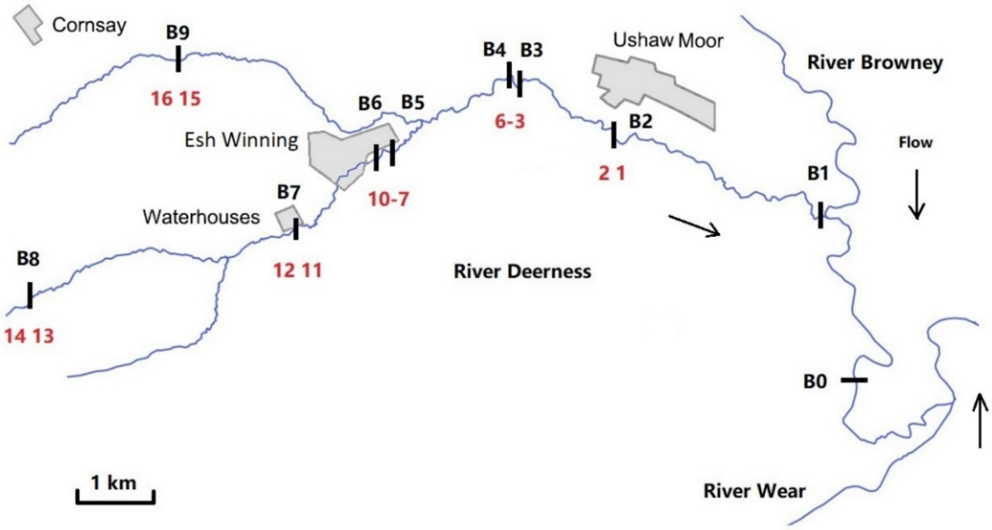


Figure 5.2 Deerness sub-catchment, location of each in-stream barrier (black) and electro-fishing sampling sites (red). Sample sites were in pairs, immediately below and above each barrier. Urban areas close to the stream are shaded grey. The bifurcating tributaries upstream of Waterhouses extend several km further upstream.

In order to improve river connectivity in the Deerness and improve passage for fish, connectivity restoration works totalling £0.5 million (including design and planning) were conducted by the Wear Rivers Trust between 2012 and 2014 in the River Deerness as part of the DEFRA Catchment Restoration Fund (Defra, 2013). Between 2012 and 2013, eight in-stream structures were identified as major barriers to fish movement in the Deerness sub-catchment. Initially B1 (shallow water at a bridge apron) was not identified as a substantive barrier and was not included in the original barrier list, but was added later in 2015. Apart from B1 (Figure 5.2), details of the remaining barriers have been described in detail in previous studies (Figure 5.3; Tummers, 2016; Tummers *et al.*, 2016) but are summarized below.

The most downstream barrier B1 is a 0.2 m high, channel wide bridge apron with ~5cm shallow water depth above it. Two baffles were installed at the upstream end of the bridge apron in 2015, in order to lift the water level on the apron surface to aid fish passage. In addition, a part-channel-width wooden baffle easement was installed on the left bank under the bridge, to provide another passage route for fish to ascend. B2 is a 1.6 m high bridge apron with five steps. It has a 30 degree gradient, and a rock ramp fish pass was installed on the right side of the apron in October 2013 (Figure 5.3). B3 is a 1.4 m high weir with an overall 35 degree gradient (though part is vertical). It was originally intended to remove this weir but, due to the presence of a gas mains pipe running across the stream channel a few tens of metres upstream, this was not possible, due to the risk of damage from bed incision. Instead, a 1.6 degree slope nature-like bypass, taking all of the flow at ~Q50, was installed on the left bank in October 2013 (Figure 5.3). B4 is a road crossing formed from seven pipe culverts (also termed a pipe bridge, Figure 5.3). It has a 0.2 m head drop at the outlet, and the pipes have a three degree gradient. Each culvert has a 0.5m diameter, and length of 4 m. A 0.9 m depth pool has formed below the structure. B4 remained unaltered throughout the study period. B5 and B6 were both series of pipe culverts, 4 m long x 0.9 m diameter, two degrees gradient, and 0.1 m head drop at the outlet. Both culverts were replaced with single span bridges in April 2014.

B7 was a series of pipe culverts, 3.4 m long and 0.6 m in diameter, two degrees gradient, and 0.1 m head drop at the outlet. This barrier was replaced with single span bridge in August 2014. B8 (location, Figure 5.2) is a two-hole pipe culvert, each 0.8 m in diameter and 12 m long, with a gradient of four degrees and a 0.3 m head drop at the outlet. No restoration was conducted on this barrier as it provided a sole farm access route, without an alternative temporary access. B9 was a 28 m long single corrugated culvert, with 0.3 m vertical head drop at the outlet and a one degree gradient along the pipe (Figure 5.3). A rock pool easement with four pools was installed at the downstream end of the culvert in October 2012 to raise the downstream water level so that it flooded into the lower part of the culvert and submerged the step (Figure 5.3).

Figure 5.3 Examples of connectivity-restored barriers in the River Deerness. Left panel: before restoration; right panel: after restoration. Modified from Tummers (2016).

B0, is located on the lower river Browney (Figure 5.2), into which the Deerness runs, before joining the Wear. It was built in 1954 to gauge river flow and is still used for that purpose. It is an 18-m wide compound, 0.7m high broad-crested weir, with a 3 m long, 1.7 degree apron and a vertical truncation at the downstream end. In 1996 the weir was modified to produce a central low-flow weir crest, and two ~0.25 m high pre-impoundments were installed 29 and 16 m downstream, to raise the tailwater level and facilitate upstream passage by leaping salmonids. Tummers *et al*. (2016) provided evidence that the weir was a substantial passage impediment at low flows. A 7.1 degree Larinier bottom baffle fish pass was installed in 2017, and an elver pass in 2006 (Lothian *et al.*, 2020). The Environment Agency stocked the River Deerness and Browney with several thousand young grayling (*Thymallus thymallus*) in 2014 and 2017 to help the process of natural recovery. Otherwise there is no evidence that the Deerness has been stocked with fish in recent decades.

5.2.2 Brancepeth Beck

Brancepeth Beck is a small lowland second order (Strahler stream order) tributary of the River Wear, joining in the middle reaches (Figure 5.4). The length of the stream is 9.7 km, elevation of the source 228 m above sea level, mean gradient 19 m per km, and the catchment covers 16.9 km^2. The surrounding land use is mainly agricultural, and large parts of the riparian zone comprise a mixture of broad-leaved and coniferous trees (Tummers, 2016). The land in the upper reach is managed as Middles Plantation, Stockley Gill Plantation and Stockley Fell Plantation, with coniferous trees grown since the 1850s. The village of Brancepeth developed along the stream's middle reach since the 1850s. The Page Bank Colliery (also known as South Brancepeth Colliery) was located about 500 m west of lower Brancepeth beck, the colliery was opened in 1855 and closed in July 1931 (Durham Mining Museum, 2020). Two railways crossed the stream, the first one constructed in the 1850s and the second constructed in the 1930s, both now disused. The stream runs through a golf course in its middle reaches which was designed in 1924. Between the 1930s and 1960s, several weirs were constructed in the golf course controlled reach. The lower reach of the stream is surrounded by intensively farmed land.

The dominant substrate of the upper and middle parts of the beck is gravel/cobble and in-stream habitat has been considered to be generally of high quality for upland stream fishes (S. Hudson, pers. comm.), although like many County Durham streams it can be susceptible to low summer flows (M. Lucas, pers. comm.). The chemical status of the beck was classified as good by the EA between 2013 and 2016, but the ecological status of the beck was classified as moderate or poor under WFD, due to the poor status of the fish (Environment Agency, 2020a). The EA have a historical fish survey site at the bottom of the beck (S1), surveys were carried out with single pass electro-fishing in 2007 and 2013. The dominant species of S1 were stone loach (minimum density, 3333 per 100m^2 in

2013) and common minnow (*Phoxinus phoxinus*) (minimum density, 2000 per 100m^2 in 2013). Brown trout were present at a low density (minimum density, 10.7 per 100m^2 in 2013). Eel, perch (*Perca fluviatilis*) and brook lamprey (*Lampetra planeri*) were occasionally caught during EA electro-fishing surveys.

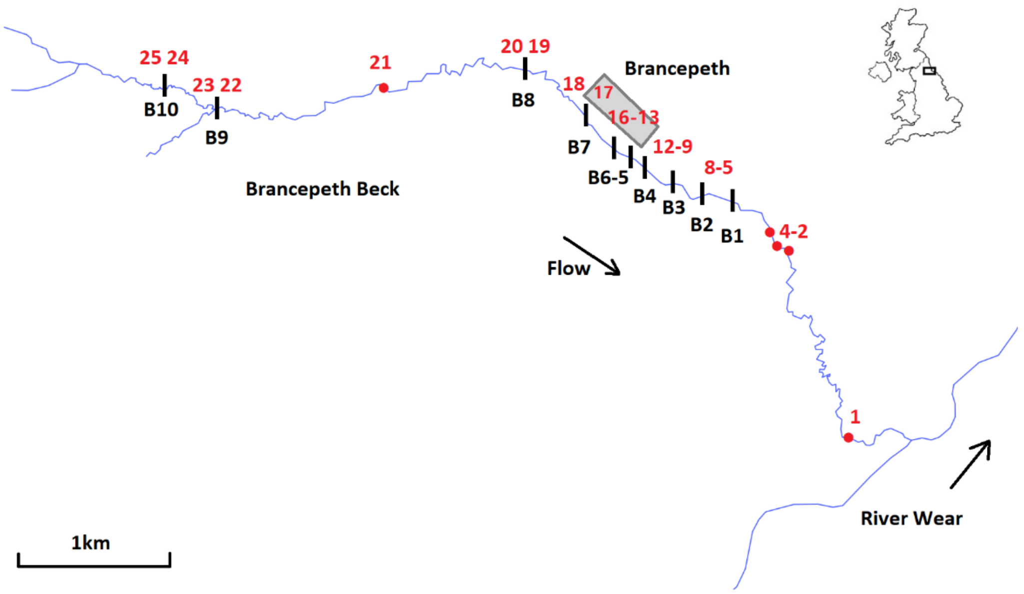


Figure 5.4 Brancepeth sub-catchment, location of each in-stream barrier (black) and electro-fishing sampling sites (red). S1-S4 and S21 are single sites. The rest of the sites (S5-S20, S22-S25) are all paired sites, one located immediately upstream and one immediately downstream of each barrier. S1 is the historical Environment Agency sampling site. Urban areas close to the stream are shaded grey.

Additionally, electrofishing fish surveys on Brancepth Beck have been carried out by the Wear Rivers Trust (WRT) and Durham University since 2014 and these have found quite low fish densities (an average of 40.9 per 100m^2 in 2014, all species combined) in the middle and upper parts of the stream (Tummers, 2016). It has been suggested that the reason for low fish densities in the stream was probably caused by the cumulative effect of multiple barriers which prevent both salmonid and non-salmonid species from moving upstream to spawn or to recolonize areas (Tummers, 2016).

Ten in-stream barriers were identified by the WRT and Durham University along Brancepeth Beck in the previous research (Tummers, 2016); the majority of these barriers were located in the middle reach of the beck on Brancepeth golf course (Figure 5.4).

These obstacles are believed to have caused significant blockage of fish to the upper catchment. In order to mitigate obstacle effects, to date six barriers have been modified by WRT with fish passage easements (B1, B2, B3, B4, B7 and B8). None have been removed and the budget for the restoration referred to has been relatively small (£65,775). Currently, there is no restoration plan for B5 and B6 due to the difficulties specific to those barriers, and the associated high costs that would be entailed. A box culvert located between B2 and B3 was mentioned in a previous study (Tummers, 2016), but it was excluded from this study, because the culvert is full-stream width and regarded as passable at most flows by all fish species (J. Sun, pers. obs.).

Barrier B1 is a 3-m long, 2.5 m diameter culvert, which had a vertical step of 0.2 m at the downstream end (Figure 5.4). In March 2016, a 3-m wide, 3.5-m long rock ramp was built immediately downstream of the culvert in order to raise the water level to provide upstream passage access through the culvert. Boulders with an approximate diameter of 1000 mm diameter were used to build the ramp revetment, prevent substrate movement and also provide sufficient water depth for fish to access. B2 consists of two bridge aprons, of which the downstream one is 0.21 m high and the upstream one is 0.41 m high (Figure 5.5). In order to remediate connectivity at the site, the downstream apron was notched in March 2016, providing an entrance for fish (primarily designed for salmonids) to access. A 5-m long rock ramp was built between the two aprons, using 1000 mm boulders as revetment to stabilise the substrate and protect the bank. B3 ("10th hole weir") is a stepped weir which consists of 10 steps, each step with an average height of 0.27 m. The weir was effectively impassable for all fish species (Figure 5.5). For restoring river connectivity at this point, the whole weir and its scour zone was filled with crushed rock and then 600 mm diameter boulders were embedded in this matrix in March 2016. A 16% gradient snaking channel over a 6.7-m horizontal section was built through the centre of the rock ramp to provide passage for fish to ascend. B4 is a 0.7-m high stone weir constructed on a 0.7-m high cascade. Removal was deemed not to be feasible so plastic baffles were installed immediately downstream of the weir in October 2016, in order to elevate depth downstream and make the weir more passable to salmonids.

Figure 5.5 Examples of connectivity-restored barriers in Brancepeth Beck. B2 before restoration (a) and after restoration (b); B3 before restoration (c) and after restoration (d); B8 before restoration (e) and after restoration (f).

Barrier B5 is a 1.5-m high concrete weir with a 24 degree slope and has not yet been mitigated. B6 is a 1.4-m high vertical weir under a road bridge, again unmitigated. B7 (owl's eyes bridge) is a 110-m long two-hole road culvert, with a two degree gradient and two aprons immediately downstream of the culvert. The height of the first apron is 0.5 m, and the second has a step height of 0.1 m. The culvert also has a 0.1-m head drop. The vertical steps to the aprons and the shallow flow through the culverts themselves made this structure impassable to fish. Plastic baulks were installed at the edge of each apron in October 2016, to improve water depth over the apron for fish attempting to pass. A notch was cut in the middle of the apron baulk, to create a water flow over the weir for jumping (salmonid) fish to access. However, the pre-barrages at B4 and B7 are only intended for jumping (salmonid) fish to pass [for which these are likely to have generated a small improvement in passage, M. Lucas, pers. comm.] and unlikely to facilitate passage for other fish species. B8 (Goodwell Ford, Figure 5.4) is a series of pipe culverts, each 0.4 m in diameter, underneath the ford, 4-m long, with a one degree gradient and with three 0.2 m vertical steps at the downstream end. Similar to B3, downstream of the ford its scour zone was filled with crushed rock and then 800-1200 mm diameter boulders were embedded in this matrix in October 2017. A pool was created immediately downstream of the pipe culverts, and all three steps were submerged underwater, in order to create passage for fish through the pipe culverts. B9 is a 10-m long, two degree gradient pipe culvert. The downstream end of the culvert is partly submerged under water which is 0.4 m deep. The upstream end of the culvert has a 1.2 m high metal trash grid which obstructs fish from ascending. B10 (see Figure 5.4 for location) was a concrete ford with a 5-cm gap underneath. The landowner replaced the ford with two pipe culverts in 2019, the downstream end of the culvert is just submerged under water for ~2 cm at normal flow level.

5.2.3 Cong Burn

The Cong Burn is a third order (Strahler stream order) stream, one of the major tributaries entering the lower Wear (Figure 5.6). The length of the stream is 17 km, the source is 180 m above sea level, and the mean gradient is 10 m per km. The whole catchment covers 39.4 km^2 and Twizell Burn is the main tributary of the Cong Burn. The length of Twizell Burn is 9.7 km, its source is 211 m above sea level, it has a mean gradient of 19 m per km and the sub-catchment covers 18.96 km^2. Much of the immediate riparian habitat is broadleaved woodland and the instream habitat for fish is generally good and varied. The dominant substrates of the burn are gravel and cobble, making it potentially a useful lowland spawning tributary for migratory salmonids, and potentially also for brook lamprey (though few have been recorded in recent times). Cong Burn is heavily modified by human activity (Environment Agency, 2020a). During the past one hundred years, the river was modified for industrial and flood defence purposes, which have led to river channelization and heavy engineering. However, industrial and urban impacts go back much further, due to the impacts of coal mining and related activities in the 19th Century, and the development of mills before that. Due to the mining history in the region, the geomorphology of Twizell Burn has changed a lot compared to its original status. The stream is also identified as a heavily modified water body by EA (Environment Agency, 2020a).

The Cong Burn is categorised under WFD as being poor for fish (e.g. in S2 trout density ranged from 1.0 - 38.6 per 100 m^2; eel density ranged from 0.0 – 3.1 per 100 m^2; stone loach density ranged from 0.4 – 5.6 per 100 m^2 between 2003 and 2013), based on EA fish community assessments by single pass electro-fishing. It is currently classified as having an overall moderate ecological status. Historical data on fish populations in Cong Burn are given in section 2.3.2.5. Historically the Cong Burn was badly polluted due to intensive coal mining and heavy industry in Twizell Burn (Grange Villa, Stanley etc) until the late-20th Century, with frequent high ammonia episodes and high phosphorus (see section 2.3.2.4 for available records of water quality). Two collieries (Newfield Colliery and Pelton Fell Colliery) were located within the Twizell Burn sub-catchment, and opened in 1835. Pelton Fell Colliery produced about 215,000 tons of coal in 1940. Newfield Colliery closed in 1936 and Pelton Fell Colliery closed in 1965. Major improvements in water quality at the Hustledown STW, Stanley, and at Combined Sewer Overflows seem to have resulted in much increased incidence of upstream migrating sea trout, spawning, and wider distribution of young trout since ~2012 (P. Frear, unpubl. data). Several obstacles (B4, B5, B6, B7 and B9) caused by weirs and culverts made the upstream reaches of the Cong Burn largely blocked to fish species such as trout (with a local emphasis on the ‘sea’ trout form). Poor connectivity was perceived by the EA as being one likely cause of WFD failure of the Cong Burn for fish.

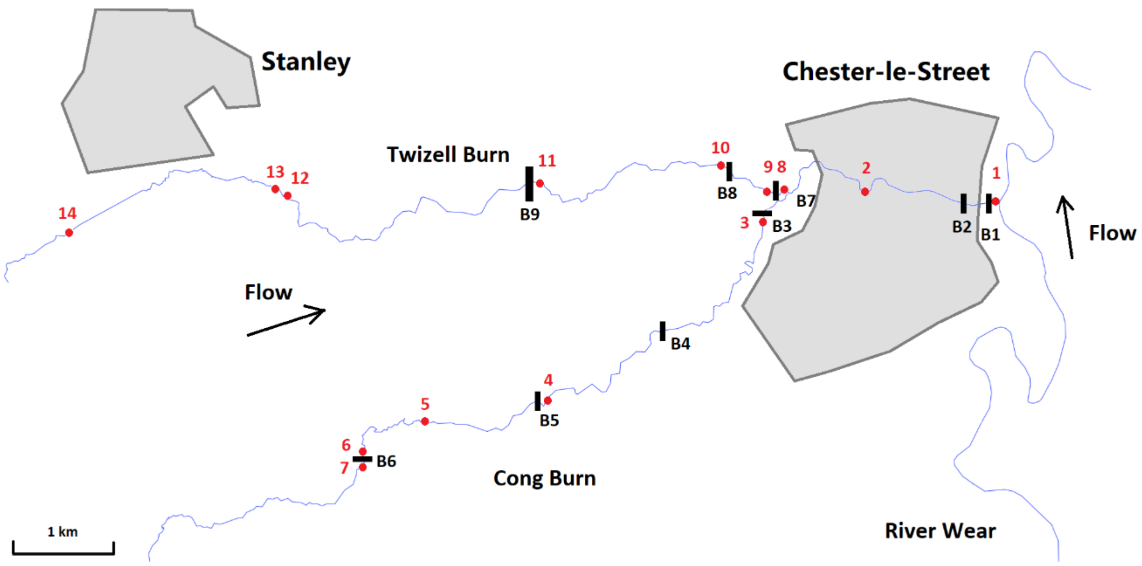


Figure 5.6 Cong Burn sub-catchment, location of each in-stream barrier (black) and electro-fishing sampling sites (red). S2, S5 and S11 were EA fish sampling sites. Urban areas close to the stream are shaded grey.

From WRT's management and restoration perspective, the Twizell Burn has been divided into three sections: the lower section is from the Cong Burn confluence to Grange Villa (B9), the middle section from Grange Villa to the Memorial Park Culvert exit and the top section from Memorial Park entrance to the source (S. Hudson, pers. comm.). The current restoration plan is to restore the longitudinal connectivity and improve the habitat of the lower section, largely because barrier B9 at Grange Villa is a large, complex barrier (see below).

In 2011, the EA, Chester-Le-Street and District Angling Club (CLSDAC) and WRT identified a number of obstructions which could have negative impacts to fish migration in the Cong Burn (Figure 5.6, 5.7). Several of these barriers could potentially be removed or altered to assist fish movement between the main river and spawning grounds (in the case of salmonids) and to facilitate recolonization by several species (e.g. bullhead, brook lamprey) lost from large parts of the subcatchment due to past pollution and poor habitat. Several of these have been carried out before and during the timescale of this thesis.

B1 is a weir located 50 m upstream from the confluence with the main river (Figure 5.6,

Figure 5.7). It belongs to Northumbrian Water Limited (NWL) and the purpose of the weir when constructed, was to carry the main sewer inlet to Chester-le-Street Sewage Treatment Works. The weir was approximately 1 m high at base flow, and of full channel width. The structure caused significant obstruction to all fish species. This interpretation was also based upon observation of fish below the obstacle and on EA fish survey data further upstream which showed low fish densities and low species richness (few fish species; see section 2.3.2.5). Because the weir could not be removed due to its proximity to the NWL sewer pipe, a fish pass was designed to mitigate the problem and assist bidirectional fish passage of the weir. A rock ramp design was chosen as being most suited to the site and for a range of species. Installation was carried out in June 2011. Free-standing large stone blocks of approximately 1000 x 500 mm x 500 mm were used to form pre-barrage check weirs, and boulders and cobbles were used to create graded banks. The rock ramp was created from four pools (length, 4 m) and small cascades (drop, 0.2 m) downstream of the weir (Figure 5.7). The rock ramp created a 1:20 slope, raising the water level to drown out the existing weir.

Figure 5.7 Examples of connectivity-restored barriers in the Cong Burn. B1 before restoration (a) and after restoration (b); B6 before restoration (c) and after restoration (d); B7 before restoration (e) and after restoration (f); B8 before removal (g) and after removal (h).

B2 is a 500 m long concrete urban culvert which was built in 1932 and begins about 200 m upstream of B1 (Figure 5.6). Most of the stream between B1 and B2 is a 200 m long concrete channel with poor habitat and high velocity during high discharge (Figure 5.8). The slope of the culvert is ~0.5 degrees and slope of the concrete channel is 0.3 degrees. The Cong Burn runs through the culvert underneath Chester-le-Street town centre and exits upstream as a heavily modified channel and remains so for several hundred metres. The concrete channel and culvert represent heavily engineered and poor aquatic habitat and they restrict the upstream movement of many fish species, partly due to a lack of habitat diversity and partly due to shallow, fast flow velocities. Due to the darkness along its length, the culvert may also act as a behavioural impediment for fish entry and passage in both upstream and downstream directions (Lucas and Baras, 2001). Durham County Council, with partners (EA and WRT) is currently working on a flood prevention scheme, and the second phase of the scheme is to open a 90 m stretch of culvert and modify the section to a more natural river reach. Part of the plan is also to restore 100 m of concrete channel to a more natural stream reach with meander and natural river bed. Neither of these were commenced during the timescale of this PhD.

Figure 5.8 Examples of unrestored barriers in the Cong Burn: (a) B2; (b) and (c) a series of weirs formed B9.

B3 was located in the Cong Burn, 300 m upstream of the Twizell Burn - Cong Burn confluence (Figure 5.6). The function of the former concrete weir and its date of construction are unknown. It posed a significant obstruction to all fish species, even at high flows, and woody debris often caused blockages upon the weir. In 2010, a weir removal project was implemented at this site. The weir was removed by CLSDAC, in partnership with the EA, in October 2010. After weir removal, the faster flows caused regrading of the river bed in the locality and generated an area of salmonid spawning habitat, though this has not been measured or evaluated for spawning use to our knowledge (S. Hudson, pers. comm.). No pre- or post-restoration habitat records in the immediate vicinity are available.

B4 is an 85-m long culvert at Waldridge Fell (Figure 5.6), which is an abandoned railway culvert with a two degree gradient. It is located 5 km upstream of the Cong Burn's confluence with the Wear. Its downstream exit has a 0.7m high cascade and it had a uniform concrete base with shallow water depth which made it difficult for salmonid fish to ascend. B5 is a 40-m long road culvert at Edmondsley, located 7 km upstream of the river mouth, also on Cong Burn and ~ 2 km upstream of B4 (Figure 5.6). Although there is a vertical step at the downstream end of the culvert, it was submerged during normal flow conditions. The gradient of the culvert is two degrees, with shallow water depth (~3 cm) on the culvert surface, which made it difficult for fish to pass through it. In order to mitigate these problems, 1800 x 210 mm concrete baffles were installed on the bed of both B4 and B5, 52 baffles were installed in the first culvert and 36 baffles were installed in the second culvert. These baffles were positioned throughout the entire length of each of the culverts in October 2010 by CLSDAC. In addition, the downstream end of cascade at B4 had baffles installed in 2019, to provide increased depth for salmonid fish to ascend (Figure 5.7). After installation, water is supposed to be held by the baffles at a depth of at least 200 mm, which should allow fish salmonids to enter and ascend the culvert even during low flows. However, at low flows the author observed no such retention of water (Figure 5.7), although they may function better at elevated flows during which most salmonid migration occurs. The baffles may also provide resting areas for fish during high flows. It seems that to date no pre-post evaluation of the efficacy of these interventions have been done by EA, CLSDAC or WRT.

B6 is a 14-m long culvert, with a gradient of two degrees (Figure 5.6). Four steps (overall height 1 m) are located immediately downstream of the culvert. It was identified by the CLSDAC as a likely obstacle to salmonid passage, due to the culvert characteristics and steps on the downstream exit of the culvert. Concrete baffles (1800 x 210 mm) were installed inside the culvert and on the exit steps by CLSDAC in August 2012, which then formed a pool with elevated water levels below the culvert, facilitating access for migratory fish (designed for salmonids). However, the baffles suffered serious damage during the spates of late autumn 2015. In summer 2019, baffles on the steps were replaced with plastic sleepers by the WRT.

B7 is Pelton Fell Bridge apron, Twizell Burn, is located 70 m upstream of the Cong Burn - Twizell Burn confluence (Figure 5.6, Figure 5.7). The apron on the downstream end of the bridge consists of three steps with a combined height of 0.6 m and a concrete base which created a barrier to fish attempting to move upstream. The channel (7 m wide x 14 m long) under the bridge, on the bridge footings, created a shallow fast flow which was also difficult for fish to pass. For the bridge apron, 2500 mm long x 230 tall mm baffles were installed on the bedrock/concrete footings of the bridge at approximately 45° angle to the bulk flow with a 2 m interval and on alternate sides of the channel (Figure 5.7), angled so as to generate a sinuous flow of water at low water levels. This was installed in May 2016 by WRT. The series of baffles is intended to provide sufficient depth and reduced flow velocities to facilitate upstream passage by several species of fish (S. Hudson, WRT pers. comm.), but the design was aimed principally at upstream passage of adult salmonids. After construction was completed, the water level immediately upstream had increased by approximately 300 mm and resulted in a small amount of impoundment.

A small rock ramp was constructed by WRT at the downstream step of Pelton Fell Bridge apron to assist upstream access for small fish species and young life stages access onto the bridge apron (Figure 5.7). The rock ramp construction was finished in 2018 spring. In addition several more baffles were placed at the downstream end of the bridge apron, to reduce the flow velocity and increase water depth on the apron. As part of this PhD study, habitat measurements and hydraulic measures were taken up to 20 m upstream and downstream of the bridge apron before and after the rock ramp construction.

B8 was Pelton Fell Weir (Figure 5.6, Figure 5.7), on Twizell Burn, located 500 m upstream of the Cong Burn confluence. The purpose of the weir was to support a foul water pipeline which crossed the beck and is now unused and abandoned. The structure was built on top of bedrock, was about 2-m high and was considered to be completely impassable to all upstream-moving fish under most flows. The most feasible restoration option was to remove the structure. The removal of the structure was felt to be low risk with regard to flooding risk locally, and there would be little adjustment in the height and gradient of the channel following removal. The weir was removed by WRT in March 2016 (Figure 5.7) to provide passage of all fish species and to reinstate natural sediment transport processes. In addition, WRT also identified a heavily modified river reach at Newfield Bridge, Twizell Burn, a few hundred meters upstream of B8. The connectivity of this river section was affected by some damaged gabion mattresses. In order to create free passage to fish, a full width channel rock ramp with four pools was built by WRT in this reach in 2018 spring.

B9 is a complex barrier, comprising a series of concrete step weirs (Grange Villa Dam) across several sites within a few hundred meters of stream length, (Figure 5.6, Figure 5.8). The total height of the structure is about 19 m, consisting of two weirs at the downstream end, a staircase in the middle, a culvert and a five stepped weir in the upstream end. The main ‘staircase’ includes 12 steps, each 1 m high, has vertical concrete edges with reinforced steel piling. The date of origin of this structure is unknown but it was built on a site with former coal mine spoil and a very unstable valley edge, and incorporates a road, under which Twizell Burn runs. Based upon the concrete structure and the timescale of redevelopment and planting trees on former pit spoil heaps, the structure likely dates from the 1960s. At the bottom of it some iron oxide (‘ochre’) leaches into the stream. This barrier is totally impassable to all fish species. For now it seems there are no plans to remove the obstructing effects of the barrier, because of its large size, and due to the complexity and expense of doing so for limited benefit in likely restorative outcomes. From Grange Villa upstream on the Twizell Burn there are several further obstacles in the form of weirs and culverts that limit fish migration and dispersal.

In the upstream reach of Twizell Burn (S14), a first order tributary called Stanley Burn, which flows through South Moor Memorial Park, was diverted into a culvert to make way for a communal paddling pool in the 1950s. Since then, the stream channel was dry with no sustained flow. The pool was used for around 15 years and then buried. In October 2017, a habitat restoration project was conducted at this site by the WRT, the paddling pool was removed and the river was diverted back to its original channel to create new habitat for fish and invertebrates.

5.2.4 Bedburn Beck

Bedburn Beck is a fourth order stream located in the middle River Wear catchment in a rural area near to the villages of Hamsterley (Figure 5.9). The source is 378 m above sea level, the length of the main stream (from Euden Beck confluence to Wear) is 8.2 km, mean gradient 9.5 m per km and the catchment covers 10.5 km^2 (Environment Agency, 2020a). The water body has no “artificial” or “heavily modified” designation, although there is an EA flow-gauging station, with no fish pass, located 1.3 km from Bedburn Beck - Wear confluence. The gauging weir is one m high, and it has a 17 degree slope. A 1.2-m deep scour-pool has been formed immediately below the weir. Several further mill weirs and other obstacles occur further upstream (Chapter 3, Barriers Database Appendix I), but Bedburn Beck has relatively few obstacles compared to the streams mentioned above, and has relatively little human development and urban habitat along it, compared to the previously mentioned streams. The stream seems to have avoided the worst of industrial and urban development activities and intensive farming. Small-scale metal mining did occur in the upper catchment, mostly before 1900. Coal mining was absent in the Bedburn catchment; the nearest coal veins mined commercially (Copley, Butterknowle) were in the upper reaches of the River Gaunless, just 2 km to the south. Water-powered fulling mills (for processing sheep wool) are recorded in 1380, and several larger mills developed on Bedburn Beck and its lower tributary, Harthope Beck by the 1800s.

Low-intensity farming (mostly livestock) is mostly in the lower catchment, while much of the middle and upper reaches are afforested. In 1927 the recently formed Forestry Commission purchased 4000 acres of land, forming what is now Hamsterley Forest, to increase national stocks of scarce timber production. In recent years Hamsterley Forest has increasingly been managed as a public amenity site, and with consideration for natural biodiversity. The river runs through forestry plantation areas, largely coniferous, but with some broad-leaved tree and pasture riparian habitat in the lower reaches. The upper catchment transitions to moorland. The river (from Euden Beck confluence to Wear) failed the WFD for its chemical status (due to elevated cadmium and lead) in 2013 due to the pollution from abandoned metal mines, which are widespread throughout the North Pennine rivers, including the Wear. However, the chemical status of the main stream achieved 'good' status in 2015, and the ecological status in 2016 was classified as moderate. Water quality data as far back as national records are available were presented in section 2.3.2.4. The predominant substrate of river is cobbles and pebbles, also with significant amounts of boulders, and large amounts of riparian cover, which provide high quality nursery habitat for salmonid fish. The river was chosen as a reference site in this study due to its relatively high quality habitat and low human impact, in terms of urban and industrial development and associated infrastructure and impacts. The fish density changes in Bedburn Beck (see section 2.3.2.5 for historic changes from EA electrofishing data) might be expected to reflect year-to-year trends in fish communities, particularly of anadromous salmonid populations, in Wear nursery streams, affected by factors such as marine survival, and river flow affecting upstream migration past barriers in the main Wear.

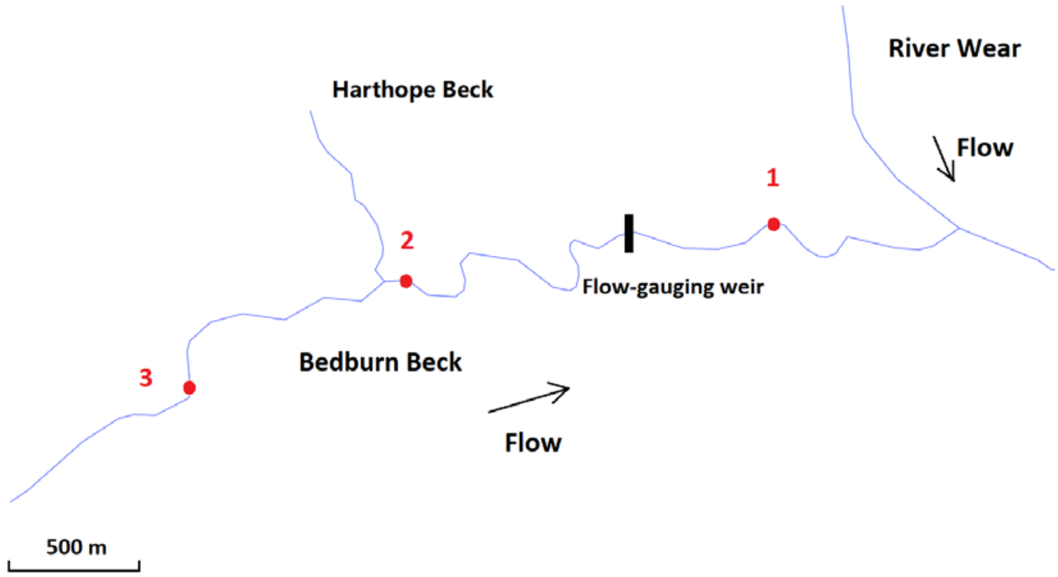


Figure 5.9 Bedburn Beck and location of the flow-gauging station (black) and electro-fishing sampling sites (red). S1 and S2 are EA fish sampling sites.

5.3 Methods

In this study, multiple fish sampling sites were selected from each stream. The length of each site varied from 50 to 80 m. Each site contained varieties of mesohabitat, including riffle, glide and pool, to increase the possibility of sampling all species of fish with different mesohabitat preferences. Fish were sampled in summer (typically July and August, when young-of-year [YoY] salmonid fry are large enough to be sampled and readily identified to species) 2017, 2018 and 2019 at the same site for each stream during base-level water flows. Fish survey data for all, or a subset of the sites, using the same methods were also available from Durham University surveys, in the Deerness (autumn 2012-summer 2016) and Brancepeth Beck (2014), as well as periodic EA and WRT sampling data at various sites on the four study streams.

The three-pass electrofishing ‘depletion’ method (Reynolds and Kolz, 2013) was carried out in the River Deerness and Brancepeth Beck. Single pass electrofishing was carried out in the Cong Burn and Bedburn Beck. This was because, for these streams, most historical EA and WRT electrofishing survey data were obtained by single-pass fishing, therefore to give comparable, standardized data for these streams, we also applied single-pass fishing method in both stream. Stopnets (4-mm mesh) were placed at the boundaries of the sampling site to delimit the fished section. Fish were sampled by electrofishing using wading with a single anode, operated with a bankside generator and control box (Honda EU10i, Electracatch WFC1, ~200 V). For triple-pass fishing, after the first and second rounds of electro-fishing, a minimum of 30 minutes were given to let the sediment settled down and allow fish to recover, and generate a relatively equal catchability between rounds, based on prior experience in these streams. Fish removed from each pass were kept in separate aerated containers, after which the catches were processed separately. Fish were identified and measured for total length. If more than 50 fish of a species were caught at a site, then 50 per species were measured at random. Fish were released back to the capture location after being processed. Capture efficiency data were calculated from raw multi-pass fishing data for each site and common species, at each stream in each year for which raw data were available and tabulated, for Durham

University data and EA data. For Bedburn Beck, capture efficiencies of salmon and trout at S2 between 2001 and 2019 were calculated based on the EA three-pass electro-fishing data. Then mean capture efficiency (2001 – 2006, 2019) was applied on the EA single pass electro-fishing year (2007-2016), to calculate the estimated fish density. The estimated absolute EA salmonid densities were only used for the salmonid density grading system in the Discussion. The Durham University (single pass) data were not used to estimate absolute fish density at Bedburn Beck sites.

5.3.1 River Deerness

A total of 16 sites were selected in the River Deerness. Paired sites were located immediately upstream and downstream reach of each barrier except B1 (Figure 5.2) which, in the Catchment Restoration Fund Deerness project (2012-2014), was not identified as a likely barrier until 2015 and so was left out of the surveying by Tummers (2016). The survey sites in this thesis remained the same as those employed by Tummers (2016), for which raw data (autumn 2012 [two sites only in autumn 2012] – summer 2015) were available. Fish population surveys carried out for this PhD were conducted in July 2017-2019 under base flow conditions, continuing earlier data series (2012-2016) by the Aquatic Animal Ecology group, Durham University, and giving a 7-year standardized data set for all sites (2013-2019). The length of each fish-sampling site varied from 60 to 80 m, depending on the spatial distribution of mesohabitats. All fish sampling was carried out under permit, issued by the Environment Agency. A BA (before-after) design was performed in this study, the status of each barrier represents ‘before-after’ (before: before restoration; after: after restoration). The sites located immediately upstream and downstream of B4 and B8 were considered as control sites, because there was no restoration work conducted on the associated barriers. Barrier B9 was mitigated in October 2012, and bullhead, minnow and stone loach were not recorded upstream prior to mitigation but colonized upstream by 2013 (Tummers, 2016; Tummers et al., 2016). Therefore, the upstream extent of distribution of bullhead, minnow and stone loach in contiguous 50-m sections was recorded by electrofishing in summer 2016-2019 to record the continued recolonization.

Although the study site distribution, across B1-B9 (S1-S16), enables BACI outcomes to be considered at the site level, it has already been argued in this chapter (section 5.1) that restoration of fish communities in obstructed rivers, may require reconnection at many barriers (due to cumulative barrier impacts). Therefore, the connectivity restoration treatment level may be argued to be at the extended reach, or whole stream level, rather than locally at barriers. In that regard, the only evidence of treatment effect at the subcatchment level that can be evaluated is the before: after response, and comparison along the same timescale as one or more reference catchments (e.g. Bedburn Beck).

5.3.2 Brancepeth Beck

A total of 25 sites were selected in Brancepeth Beck. S1-S4 were downstream reference sites, in the lower part of the stream which is not affected by any artificial barriers. S21 was used as an upstream reference site, in a locality without close proximity to barriers, which had been surveyed previously in 2014 (Tummers, 2016). The remaining sites were all paired, one located immediately upstream and one immediately downstream of each in-stream barrier. The length of each fish sampling site varied from 50 to 80 m, depending on the distances between barriers as well as the spatial distribution of mesohabitats. Three pass electro-fishing surveys were conducted at each site between July and September under base flow conditions. All sites were surveyed in 2017, 2018 and 2019. Pre-intervention electrofishing raw data for S4-S24 in 2014 were available from Tummers (2016). In addition, redd (salmonid spawning nest) counting surveys were conducted through the river (between S1 and S25) in 2017 and 2018 in late November, to check the upstream penetration distance of adult sea trout (no salmon recorded in this stream) and the level of their spawning activity. These were performed by slowly walking the bank in an upstream direction, when water was sufficiently clear, looking for newly generated spawning redds and large adult trout. Trout spawning begins in early November, and ends in December in this and nearby streams, so the timing optimized observation of newly generated redds before they could be modified by gravel movement during spates.

5.3.3 Cong Burn

A total of 14 sites were surveyed for fish in Cong Burn. S1-S11 were located within the connectivity restored area. S12 and S13 were located immediately upstream and downstream of a Northumbrian Water Sewage Treatment Works (SWT) outflow. S14 was located within the Twizell Burn restoration reach. S1, S2 and S5 were previous EA fish sampling sites for which historic data are available, and the remaining sites were previous WRT fish sampling sites. The WRT conducted yearly (summer) single pass electro-fishing surveys (L = 30 m) for S3 and S4 from 2011 to 2016; as well as for S6, S7 and S10 from 2013 to 2016. However, the rest of the sites have not been surveyed across successive years. EA surveyed S2 with single pass electro-fishing in 2003, 2007, 2009 and 2013; S2 with three pass electro-fishing in 2009, 2011, 2013 and 2015; S5 with single pass electro-fishing in 2003, 2007, 2011, 2012 and S11 with single pass electro-fishing in 2013 (see section 2.3.2.5). No surveys were conducted at many of the Cong Burn / Twizell Burn sites before 2010, which means there is no baseline fish population data for most of those connectivity restoration sites on the Cong Burn catchment.

The length of each fish sampling site surveyed as part of this PhD varied between 50 and 80 m, depending on the spatial distribution of mesohabitats. Single pass electro-fishing surveys were conducted at each site between July and August under base flow conditions. All sites were surveyed in 2017, 2018 and 2019.

Apart from electro-fishing, a detailed habitat survey was performed at Pelton Fell Bridge (B7) and Newfield Bridge, to assess the changes in river habitat before and after the rock ramp installation. Wetted width, depth (25%, 50% and 75% quartiles) and flow velocity (25%, 50% and 75% quartiles; 50% water depth) were measured every 1 m in both upstream and downstream section (L= 20 m) before and after construction. These data were used for drawing the 2D graph, to visualize the habitat changes post the restoration. The survey was carried out in February 2018 before the restoration and June 2018 after restoration under base flow conditions.

5.3.4 Bedburn Beck

A total of three sites were selected in Bedburn Beck. S1 is located 0.6 km downstream of the flow-gauging station. S2 is located 1.2 km upstream of the flow-gauging station and

S3 located 1 km further upstream of S2 (Figure 5.9). The length of each fish sampling site varied between 50 and 60 m, depending on the spatial distribution of mesohabitats. Single pass electro-fishing surveys were conducted at each site between July and August under base flow conditions. All sites were surveyed in 2017, 2018 and 2019.

In addition, S2 is an EA fish 'index site' and has been surveyed between 1995 and 2014 across most years, with salmonid only surveys in 1991 (section 2.3.2.5). The site was surveyed by the EA with single-pass electro-fishing method in 1995, 1997, 2007-2012 and 2014; and the three-pass method in 1991 and from 2001 to 2006, 2019. The first run fish abundance data from the three-pass survey (2001 - 2006) was used to calculate the minimum fish density, then EA minimum fish density was used in combination with the minimum fish density data from this study, to monitor the long term fish density changes in a standardised manner.

5.3.5 Data analysis

Before analysis, data were checked for normality test using conventional tests, and necessary transformations were applied when needed. For the River Deerness and Brancepeth Beck (multi-pass depletion fishing), fish densities per site were calculated based on Carle and Strub's K-pass removal method, with the R (version 3.6.1) package 'FSA' (Ogle, 2020). Total fish densities were calculated by summing the densities per species (from Carle and Strub) thereby accounting for differing catchability byetween species. Fish densities data were fourth-root transformed (Boys *et al.*, 2012), and fish length data were log 10 transformed to meet assumptions of normality before conducting analysis. Permutational multivariate analysis of variance (PERMANOVA) was used to determine changes in the fish communities through the time course of and following the restoration work, and the differences between paired upstream and downstream sites, using the R 'Vegan' package (Oksanen *et al.*, 2019). If significant differences in fish communities were found, a similarity percentage (SIMPER) analyse based on the decomposition of Bray-Curtis dissimilarity index (Clarke, 1993) was followed to identify which species contribute more dissimilarity in fish abundance among study sites. Linear Mixed Modelling (LMM) was performed to analyse the changes in fish density using the ‘lme4’ and ‘lmerTest’ package (Kuznetsova *et al.*, 2017), with fish densities as fixed factors and sites as random factors. The post-hoc Tukey's multiple comparison test was performed to analyse the differences in abundance of each species and total fish density between each study site, using 'multcomp' package (Hothor *et al.*, 2020).

5.3.5.1 Data analysis - River Deerness

The River Deerness female trout spawning population is dominated by sea trout, with a mixed contribution of male spawners from sea trout, freshwater-resident brown trout and precocious parr [= sexually mature trout in the ‘juvenile’ parr form] (Tummers, 2016; Winter *et al.*, 2016; Lothian *et al.*, 2020). Many of the freshwater-resident adult trout in the Deerness migrate upstream from the rivers Wear and Browney to spawn (Lothian et al. 2020). This results in a trout population age structure in the Deerness dominated by young juvenile trout; most emigrate as smolts to become sea trout, and some emigrate to the main river to grow as freshwater-resident trout. Based on frequency distributions of lengths it becomes relatively easy to distinguish YoY (‘fry’) from older parr and small adults. Therefore in the River Deerness, trout sampled were split into Young-of-the-Year (YoY) trout (age 0+) and older trout (age 1+ and older) based on the length frequency distribution (YoY: Year 2013-17, length < 90 mm; Year 2018-19, length < 80 mm; Older: Year 2013-17, length ≥ 90 mm; Year 2018-19, length ≥ 80 mm). The different size cut-offs between years, reflect different growth rates between years. This enabled year-to-year fluctuations in recruitment to be analysed. Kruskal-Wallis *H* Tests were used to compare the difference in median trout length (irrespective of age) through the study periods. Mann-Whitney *U* Tests were used to compare the trout densities in each age group between the paired study sites in each year.

Spearman's Rank Correlation Coefficient was used to determine whether there was any correlation between YoY trout density and the mean flow (m^3s^{-1}) in the previous autumn / winter (1 September to 15 December) as well as the high flow event days (number of days with flows exceeding Q5 [upper 5%-ile of flows] or Q10 or Q20). Because autumn flows affect the ability of salmonid spawners to access streams (Svendsen *et al.*, 2004), it was hypothesized that high flows would bring more adults into spawning streams, resulting in greater egg deposition and more YoY in the next summer.

In addition, the correlation between YoY trout density and the frequency of high flow event days (number of days with flows exceeding Q1 or Q5) in winter / spring between the end of spawning and estimated fry swim-up from the gravel (estimated as 16 December and 15 May) in the same sampling year was examined. This was because after the spawning season, very high flow events (flow exceeding Q1 or Q5) could wash gravels out (and eggs / alevins with them), and lead to a reduction of YoY survival rate (Jensen and Johnsen, 1999; Hendry and Cragg-Hine, 2003). So high flow events in spring may negatively relate to summer YoY density in the same year. The daily mean flow data between 2012 and 2019 were gathered from the Environment Agency flow gauge station at B0, which is the closest to the study site, highly correlated with Deerness flow (Lothian, 2021).

5.3.5.2 Data analysis - Brancepeth Beck

In order to assess the changes in fish community before and after the period of connectivity restoration, fish density data from several previous surveys were used. These comprised S4-S24 three pass electro-fishing data, from surveys by Durham University in 2014 (Tummers, 2016); S1, S7, S14, S17, S19 and S20 single pass electro-fishing data, from surveys carried out by the WRT in 2015. S5-S14 and S17-S20 single pass electro-fishing data, survey carried out by the WRT in 2016. Because the WRT never conducted double or triple pass electro-fishing in this stream, it was not possible to calculate the capture efficiency data from WRT fishing data. Minimum fish density was calculated from the WRT single pass electro-fishing data. To combine with the WRT single pass fish density data, the first run data of three-pass surveys from Durham University were extracted to calculate the minimum fish density then analysis was conducted. Notice, the 2014 fish densities data in this thesis differed from those in Tummers (2016), because Tummers (2016) wrongly calculated the Brancepeth Beck 2014 fish densities in his thesis (Tummers' Deerness data were unaffected). Although he states Carle and Strub's K-pass removal method was applied, for Brancepeth Beck, fish numbers from the three passes were added together, then divided by the surveyed area. Therefore I have re-calculated the 2014 Brancepeth Beck fish densities, based on raw electro-fishing data, with Carle and Strub's K-pass removal method.

A BACI (before-after-treatment-control) design was performed in this study, the status of each barrier represents 'before-after' (before: Year 2014, 2015, 2016; after: Year 2017, 2018 and 2019). Sites below all obstacles (S1-S4) were regarded as reference sites without intervention (controls), those within the golf course restored zone (S5-S12) were considered as impact sites at which connectivity restoration occurred, and sites located upstream of B4 (S13-S25) were considered as control sites at which no restoration intervention occurred, even though minor inteventions were made at B7, B8 and B10. It is recognised that there is a natural environmental cline along Brancepeth Beck, as distance from confluence (main source of colonist fish species) increases, and stream width decreases, with increasing distance from the confluence. However, stream mesohabitat conditions (riparian cover, dominant substrate, depth, flow types) remained relatively similar along the stream's course until the upstream-most sites, which were generally shallower and slower than those downstream. This was the most viable reach-level analytical design considered possible in this stream.

As for Brancepeth Beck, in order to interpret the restoration efforts on trout population recruitment patterns specifically, the sampled trout were spilt into two groups by length, YoY trout (age 0+) and older trout (age 1+ and older) based on the length frequency distribution (YoY: Year 2017, length < 90 mm; Year 2018-19, length <80 mm; Older: Year 2017, length ≥ 90 mm; Year 2018-19, length ≥ 80 mm). Because fish length was not measured in the 2014 survey (Tummers, 2016), it was not possible to compare the 0+ and older trout density changes between 2014 and other years. The fish length data from the WRT survey (2015) were extracted and used to compare with the post-restoration fish length data. Fish length data across years and sections were analysed with Kruskal-Wallis *H* Tests.

PERMANOVA was used to compare the fish communities change following the restoration work. In order to create a balanced design for PERMANOVA analysis, fish communities from S5-S12 (impact) were used against fish communities from S13-S20 (control). The remaining statistical methods followed those for the River Deerness study.

5.3.5.3 Data analysis - Cong Burn

The estimated minimum density of each species was calculated by dividing fish abundance in the single fishings carried out, by the surveyed area. Lack of good baseline data (EA data for a few sites and years, and WRT data for a few sites and years) and variability in methods (see section 5.3.3) limits the robustness of interpretation. Therefore the analysis is mostly focused on the spatial and temporal variation in fish density, and species occurrence, before and after connectivity restoration at specific obstacles and more generally over the period in which progressive connectivity restoration has occurred (2010 onwards). To interpret the restoration efforts on the trout population specifically, the trout were split into two groups, YoY trout (0+) and older trout (1+ and older) based on the length frequency distribution (YoY: length < 80 mm; older: length ≥ 80 mm). The remaining statistical methods followed those described earlier for the River Deerness.

5.3.5.4 Data analysis - Bedburn Beck

To interpret changes in recruitment of brown trout and salmon populations over the study timescale, both species were split into two groups, YoY fish (0+) and older fish (1+ and older) based on the length frequency distribution (YoY: length < 80 mm; older: length ≥ 80 mm). Spearman's rank-order correlation was used to determine the relationship between pairs of brown trout, Atlantic salmon and bullhead abundance over the study period. Linear regression was used to determine the long-term trend of YoY / older brown trout and salmon density across years at S2. Older trout and salmon densities were log transformed to meet the assumption of normality. In addition, Spearman's correlation was used to determine whether correlations existed between YoY/parr trout/salmon density and the mean flow (m^3/s) in the previous autumn / winter (from 1 September to 15 December) as well as the number of high flow event days (number of days with flows exceeding Q5 or Q10 or Q20). The correlation between YoY/parr trout/salmon density and the number of high flow event days (number of days with flows exceeding Q1 or Q5) in winter / spring (from 16 December to 15 May) in the same sampling year was also examined. The rationale for this is explained in section 5.3.5.1. The daily mean flow data at Bedburn Beck between 1991 and 2019 were gathered from the EA's Bedburn Beck gauging station for this (Figure 5.9).

Furthermore, the correlation between annual Framwellgate fish counter data and mean daily flow (m^3/s, 1 June-30 November), as well as the number of high flow event days (exceeding Q5 or Q10 or Q20) from 1 June to 30 November (representing the main period of salmon and sea trout upstream migration in the Wear, upstream of Durham based on fish counter data) was examined. The daily mean flow data between 1995 and 2019 were gathered from the Environment Agency gauging station on the Wear at Chester le Street (Durham has no discharge gauging station records). The purpose of this was to establish whether increased river flows led to increased upriver migration of adult anadromous salmonids, potentially facilitating access back to spawning tributaries such as Bedburn Beck and the River Deerness.

5.4 Results

5.4.1 River Deerness

Eight species were caught during the survey period, and the predominant species before the restoration (i.e. fish surveys in summer 2013) was brown trout (Figure 5.10, 5.11). Atlantic salmon was not recorded at any site in the Deerness in any year, despite the connectivity restoration. European bullhead and common minnow were present at most sites, typically at similar or slightly lower abundance than brown trout (Figure 5.10, Figure 5.11). Stone loach were present at most sites, but usually at lower abundance than trout, bullhead and minnow. European eel and grayling were both present at a very low density, with the occurrence and abundance of eel increasingly slightly during the study, particularly at downstream-most survey sites (Figure 5.10, 5.11). Grayling were excluded from the following analysis due to the low abundance (<0.01% of total fish caught). Since 2016, three-spined stickleback (*Gasterosteus aculeatus*) appeared in the study sites, particularly in downstream sites. A non-native Koi carp (*Cyprinus carpio*), an ornamental form of common carp, was caught at S2 in summer 2017. Because it is the only record of this species in the catchment, it was also excluded from analysis. S14 (barrier control site at the top of the catchment) was inhabited by brown trout only over the full duration of study, even though up to five species were recorded 100 m downstream at S13 (Figure 5.10). It should be noted that the number of species at S13 declined to just two species, brown trout and bullhead in 2017, 2018 and 2019. Indeed a general decline in the abundance of stone loach at sites upstream of S6 and minor at sites upstream of S11 appears evident since 2016 (Figures 5.10, 5.11).

Capture efficiencies were calculated for three types of fish species: solitary midwater (trout), solitary benthic cryptic (bullhead) and schooling midwater (minnow). The mean estimated efficiency (all sites combined) of three-pass fishing ranged from 87.4% to 92.8% for YoY trout; 91.9% to 96.9% for older trout; 78.6% to 89.8% for bullhead and 82.0% to 94.9% for minnow in the Deerness between 2016 and 2019 (Table s5.1).

The greatest species richness tended to occur at the downstream-most survey sites in the system over the study period (2013-2019), and reduced further upstream (Figure 5.10, Figure 5.11). Over the same timescale the number of species recorded tended to remain stable at those downstream sites, but reduced markedly in upstream sites from 2017 onwards, particularly upstream of S6. From 2016 to 2019, the densities of brown trout and bullhead became similar suggesting the river was dominated by both species.

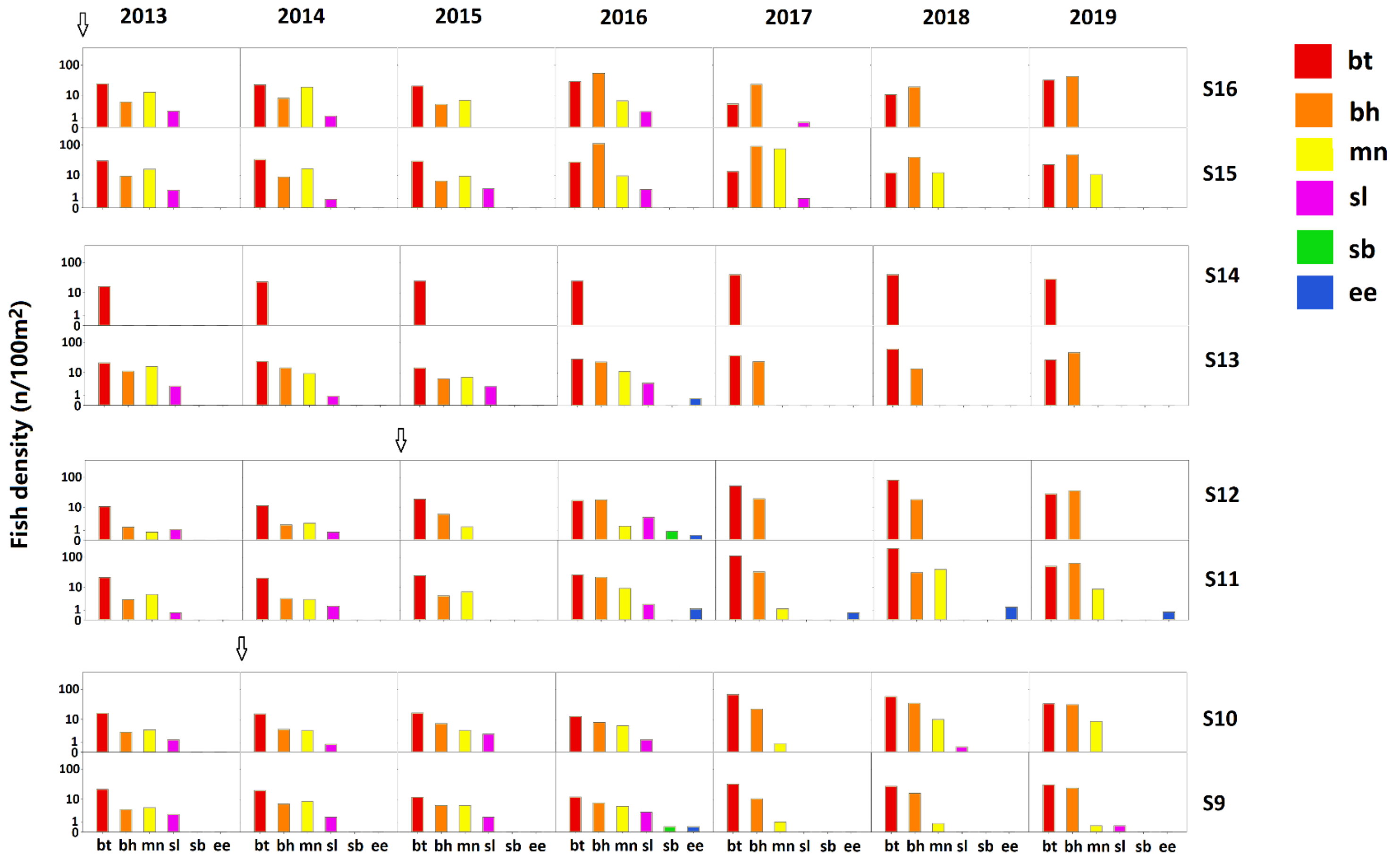


Figure 5.10 Fish density of each species (note log scale) between S9 and S16 during the sampling periods. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel. Pre-restoration data for S15 and S16 data are available for 2012, but are not presented for space reasons (see Tummers, 2016; Tummers et al., 2016). The arrows indicate the year of restoration, relative to timing of summer surveys.

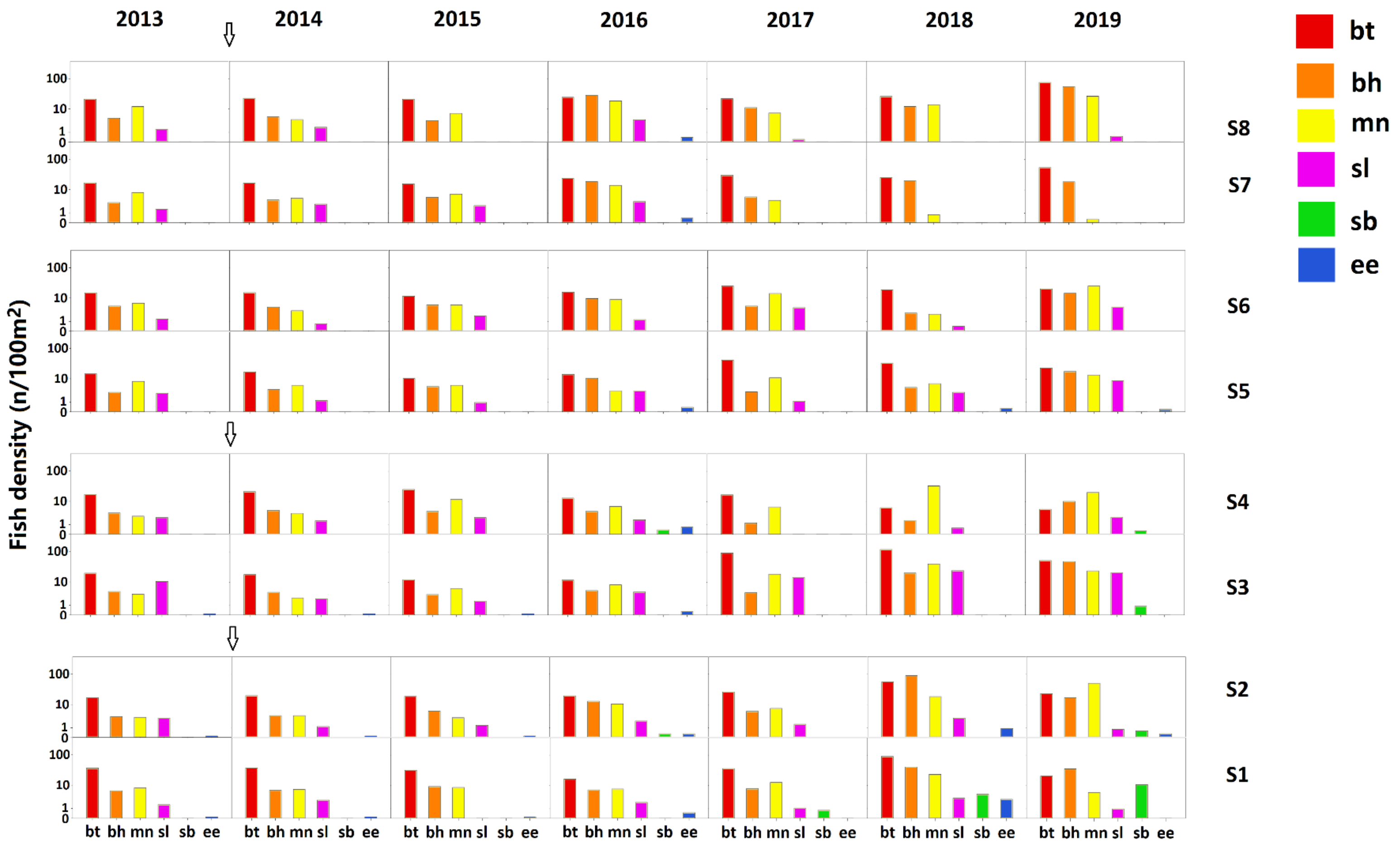


Figure 5.11 Fish density of each species (note log scale) between S1 and S8 during the sampling periods. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel. The arrows indicate year of connectivity restoration at barriers relative to timing of summer surveys.

During the study period, no significant differences were found in the fish community at sites across the catchment between the pre-intervention year (2013) and the remaining years (PERMANOVA, $P > 0.05$ in all cases). For all restored sites, no significant differences in fish community were found between upstream and downstream reaches post restoration (PERMANOVA, $P > 0.05$ in all cases) as well as between control sites S6 and S5 at unmodified Barrier 4 (PERMANOVA, $F_{1,12} = 4.5$, $P = 0.21$). Significant differences in fish community were only observed between control sites S13 and S14 at unmodified Barrier 8 (PERMANOVA, $F_{1,12} = 36.44$, $P = 0.001$). SIMPER test output showed bullhead, minnow and stone loach contributed 90.6% dissimilarity in the fish communities between S13 and S14, these being absent from S14 over the entire study period.

Although the restoration has, apparently, had negligible benefit to the overall fish community, at the subcatchment level, the total fish (all species combined) abundance varied significantly between the sampling years (LMM, $F_{6,90} = 13.26$, $P < 0.001$). The lowest total fish density across all sites (mean and SD, 32.4 ± 8.5 per 100 m^2) occurred in 2015; and the highest density (mean ± SD: 93.0 ± 70.1 per 100m^2) occurred in 2018 (Figure 5.12 which presents box plots of medians). The overall fish densities in 2013, 2014 and 2015 were significantly lower compared with fish densities in 2017, 2018 and 2019 (paired post hoc, $P < 0.001$ in all cases); and total fish density in 2016 was significantly lower compared with fish densities in 2018 and 2019 (paired post hoc, $P = 0.02$ in both cases). Furthermore, brown trout, bullhead, stone loach, three-spined stickleback and European eel abundance changed significantly during the sampling periods (Figure 5.13, Table 5.1; LMM, $P < 0.05$ in all cases). Details of these patterns are presented below. All were increases, except for stone loach, which decreased in abundance since 2016.

Table 5.1 LMM output showing the change of fish density in the River Deerness across years. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel. Site was used as a random factor in the analysis.

Species	Mean square	*df*	*F*	*P*
Total	1.146	6,90	13.26	**<0.001**
BT	0.561	6,90	6.05	**<0.001**
BH	1.692	6,90	18.61	**<0.001**
MN	0.166	6,90	0.66	0.68
SL	1.268	6,90	6.65	**<0.001**
SB	0.193	6,90	2.64	**0.02**
EE	0.523	6,90	8.17	**<0.001**
BT YoY	0.351	6,90	2.22	**0.04**
BT Older	0.306	6,90	5.06	**<0.001**

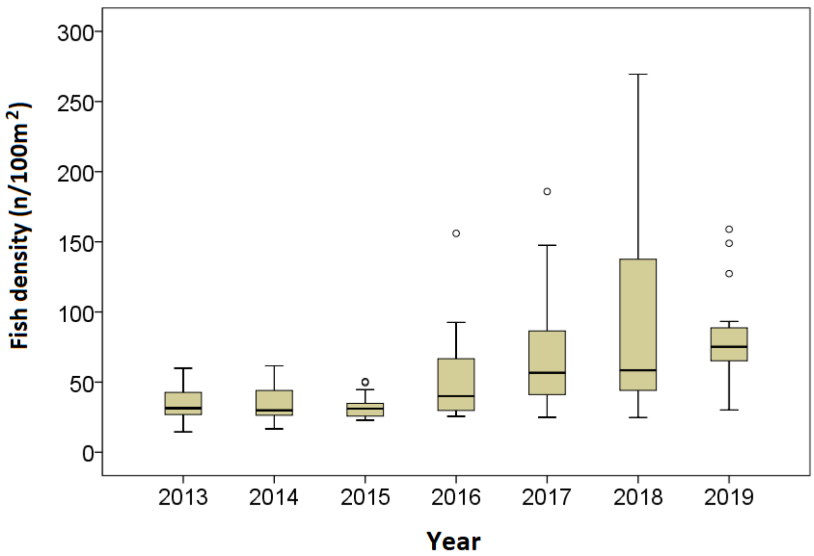


Figure 5.12 Box plots showing range, outliers, lower quartile, median, upper quartile of total fish density (per 100 m^2) in the River Deerness over the study duration.

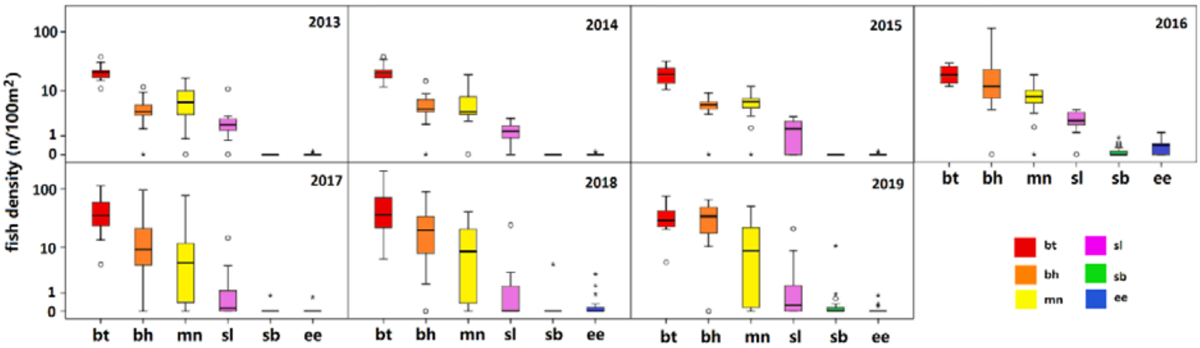


Figure 5.13 Box plots of total fish density (per 100 m^2) of each species across years for the River Deerness. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel. Note the log scale.

Similar to the overall fish density, the lowest brown trout density (19.7 ± 6.5 per 100m^2) occurred in 2015; and the highest brown trout density (53.3 ± 47.8 per 100m^2) occurred in 2018 (Table 5.2). The brown trout densities in 2013, 2014, 2015 and 2016 were significantly lower compared with the densities in 2017 and 2018 (Table 5.3). The 0+ trout density showed an increasing trend during the study period, from 10.6 ± 4.6 per 100 m^2 in 2013 to 19.8 ± 11.8 per 100 m^2 in 2019 (Figure 5.14), indicating increased reproductive success since the outset of connectivity restoration. On the other hand, the older trout density firstly showed a decreasing trend, from 10.3 ± 2.1 per 100 m^2 in 2013 to 7.5 ± 3.0 per 100 m^2 in 2016 but increased to 14.1 ± 11.3 per 100 m^2 in 2019. Overall, both 0+ trout densities and older trout densities significantly increased during the study period (LMM, YoY, P = 0.04; older, P < 0.001; Table 5.1).

Table 5.2 Mean and SD brown trout density (per 100 m^2) and average fish length (mm) across study years in the River Deerness.

	2013	2014	2015	2016	2017	2018	2019
Total	20.9±6.3	21.3±6.7	19.7±6.5	20.3±6.5	43.8±28.3	53.3±47.8	33.8±16.8
YOY	10.6±4.6	10.6±5.3	11.1±5.2	12.9±4.3	28.5±26.7	32.9±37.0	19.8±11.8
Older	10.3±2.1	10.6±2.3	8.6±5.0	7.5±3.0	15.3±10.9	20.4±13.8	14.1±11.3
Length	113.5±54.4	108.1±50.3	97.6±47.0	99.3±37.9	95.4±44.0	92.5±44.0	86.1±40.5

Table 5.3 Paired post hoc test of LMM (Tukey's multiple comparison) showing significant differences in different fish species densities in the Deerness during the study period (2013-2019).

Species	Year	*Z*	*P*
Brown trout	2013 - 2017	3.18	0.025
	2013 - 2018	3.72	0.004
	2014 - 2017	3.11	0.031
	2014 - 2018	3.65	0.005
	2015 - 2017	3.53	0.008
	2015 - 2018	4.07	< 0.001
	2016 - 2017	3.36	0.014
	2016 - 2018	3.90	0.002
Bullhead	2013 - 2018	5.62	<0.001
	2013 - 2019	8.00	<0.001
	2014 - 2018	5.09	<0.001
	2014 - 2019	7.47	<0.001
	2015 - 2018	5.01	<0.001
	2015 - 2019	7.39	<0.001
Stone loach	2016 - 2017	-4.16	<0.001
	2016 - 2018	-4.32	<0.001
	2016 - 2019	-3.69	0.004
Eel	2013 - 2016	5.29	<0.001
	2014 - 2016	5.29	<0.001
	2015 - 2016	5.29	<0.001

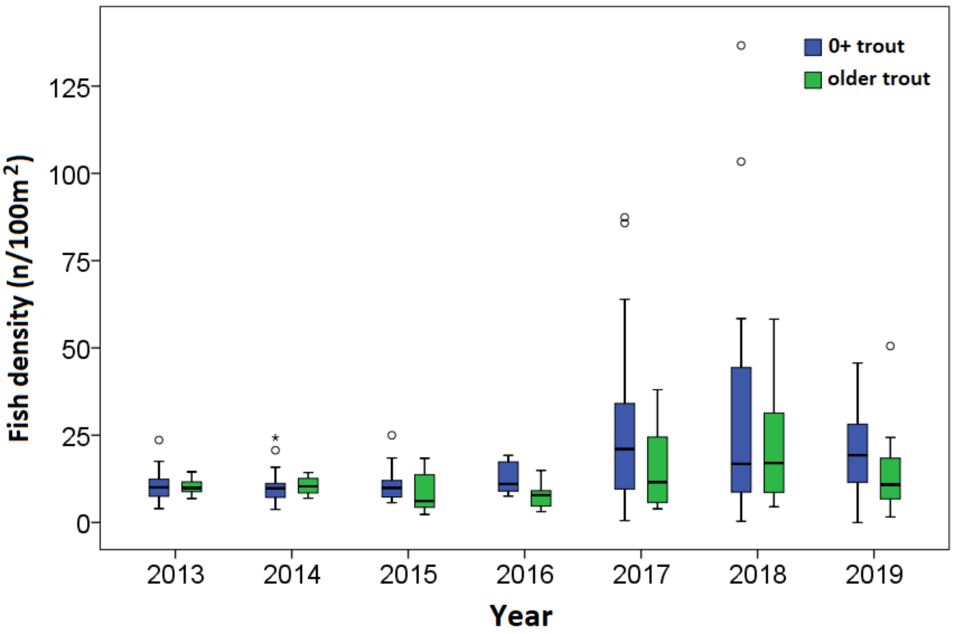


Figure 5.14 Box plots of brown trout density (per 100 m^2) in the Deerness during the study period; blue: YoY trout, green: trout of one or more years old.

Associated with the trout density, significant differences were also found in median trout length across the sampling periods (Figure 5.15; Kruskal-Wallis H Test, $X^2(6) = 332.72$, $P < 0.001$). A decreasing trend in trout length was observed. From the trout length data combined across sites, for the full study duration, a change in length frequency distribution was apparent: in 2013-2015 four clear length modes were apparent (likely equating to Ages 0+ to 3+), but from 2016-2019 only two length modes were evident, with an extended 'tail' of relatively few larger trout (Figure 5.15). This indicates that over the period 2013 to 2017 the Deerness trout population became increasingly dominated by Age 0+ and 1+ trout and with relatively fewer larger (older) trout. Within the paired upstream and downstream sites, a significant difference in 0+ trout density was found between S3 and S4 after restoration (Table s5.2; Mann-Whitney *U* Test, $U = 4$, $P = 0.025$); the 0+ trout density in the reach downstream of the nature-like bypass was significantly higher compared with the upstream impounded reach (Figure 5.16). No significant differences in 0+ trout density were found at the remaining restoration sites, including the upstream and downstream of the two unmodified structures (Table s5.2; Mann-Whitney *U* Test, $P > 0.05$ in both cases). In addition, no significant differences were found in both older trout and total trout density across all paired sites (Table s5.2; Mann-Whitney *U* Test, $P > 0.05$ in both cases). Again Figure 5.16 makes clear the increase in densities of YoY and older trout over the study period, being more evident at some sites than others.

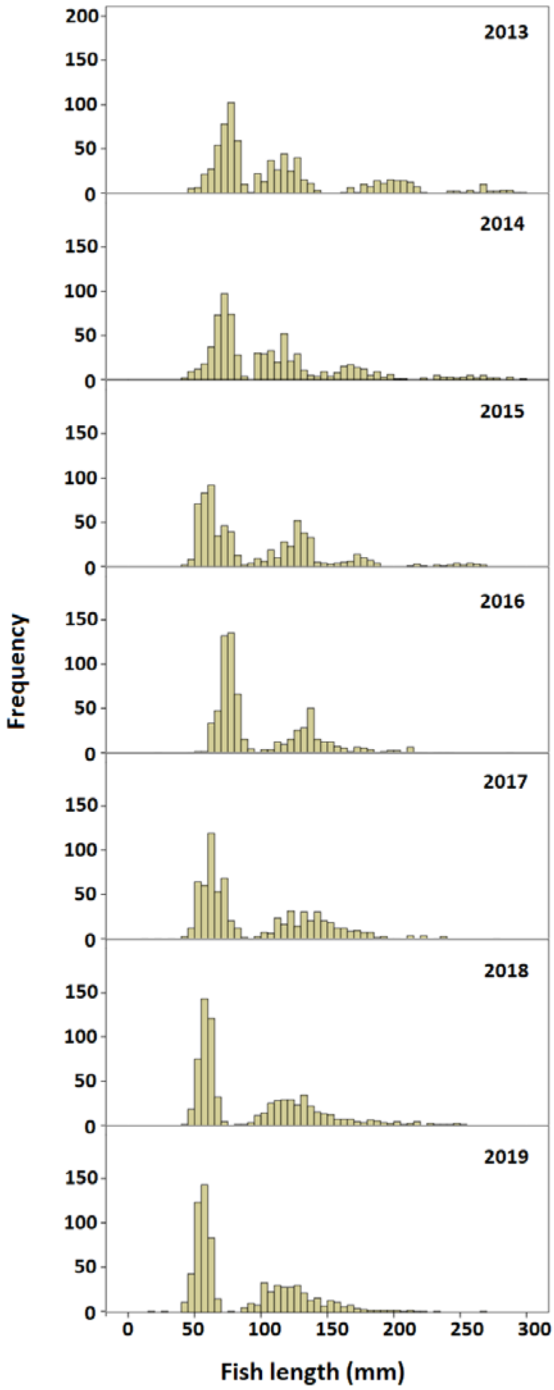


Figure 5.15 Length-frequency distribution of brown trout from the Deerness (all study sites combined) during the study period.

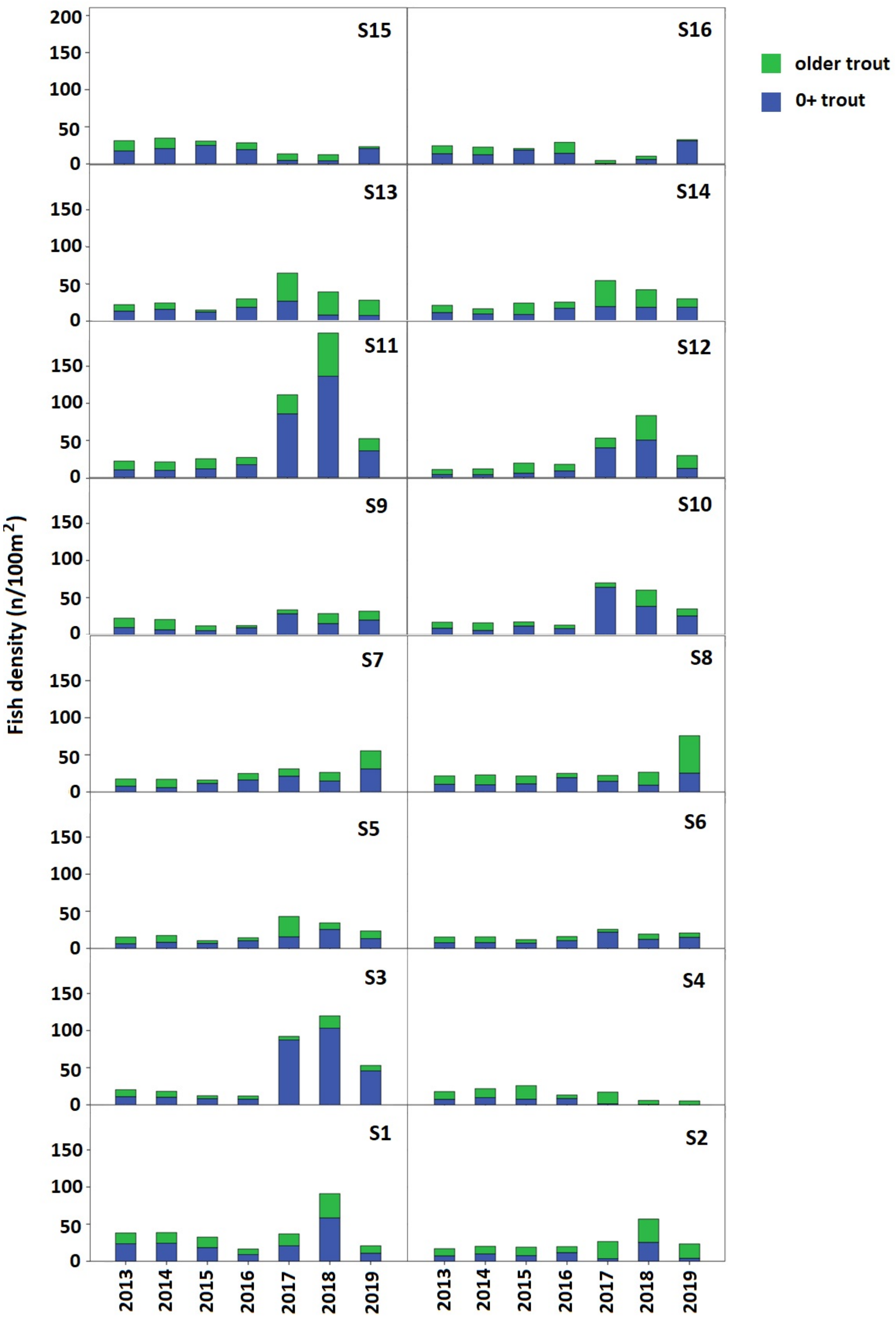


Figure 5.16 Brown trout density (per 100 m^2) in the River Deerness across the study period; blue: YoY trout, green: trout of one or more years old.

Within all connectivity-restoration sites, the 0+ trout population showed an increasing trend in the upstream sites where barriers had been removed (S8, S10 and S12). In contrast, 0+ trout in several of the upstream fish pass impounded sites (S2 and S4) were at a low abundance (except in 2018 at S2). In S4, only one YoY trout was caught in 2018 and no YoY trout were caught in 2019 (Figure 5.17). In S16 (upstream of a culvert fitted with a rock-weir fish pass), 0+ trout density did not show an increasing trend between 2012 and 2018, but the density approximately doubled in 2019 (Figure 5.17).

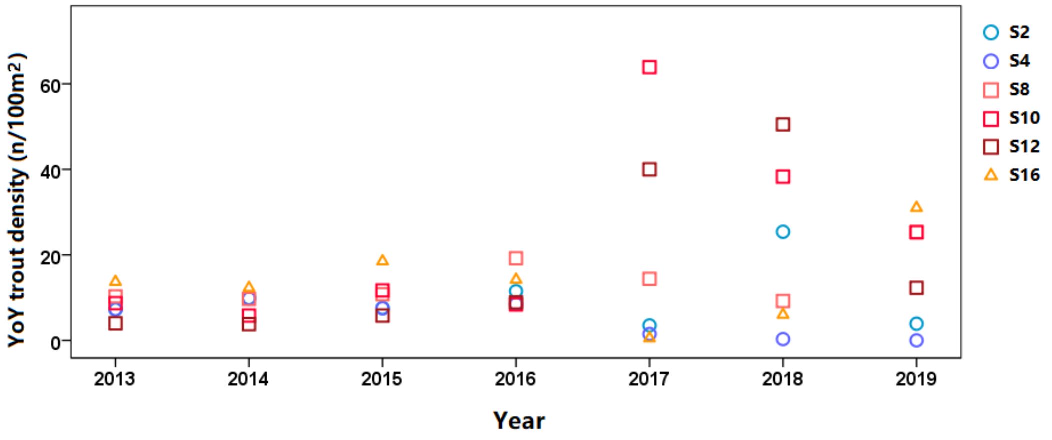


Figure 5.17 YoY trout density (per 100 m^2) changes in connectivity-restored sites of the Deerness. Circle: sites where a fish pass was installed but which retained ponding upstream of the barrier (weir, bridge footings); Square: sites immediately upstream of barrier, where the barrier was fully removed. Triangle: site at which a fish pass was installed but where there was no ponding immediately upstream of former barrier (culvert).

Mean daily flow at B0 between 2012 and 2019 is presented in Figure s5.1. A strong negative correlation was found between mean YoY trout density and high flow event days (number of days with flows exceeding Q10 between 1 September to 15 December) in the previous autumn (Figure s5.2; Spearman's Rank Correlation, r = -0.80, P = 0.031). No correlations were found between the mean YoY trout density and the mean daily autumn flow in the previous year, as well as between mean YoY trout density and number of high flow event days exceeding Q5 or Q20 in autumn (Figure s5.2; Spearman's Rank Correlation, P > 0.05 in all cases). No correlation was found between YoY trout density and the number of high flow days between 16 December - 15 May (Figure s5.2;

Spearman's Rank Correlation, *P* > 0.05 in all cases).

Bullhead was one of the most abundant species throughout the study period, their abundance increased dramatically across the majority of sampling sites since 2016 (Figure 5.11, Figure 5.13). Overall, the bullhead densities in 2013, 2014 and 2015 were significantly lower compared with those in 2018 and 2019 (Table 5.3; paired post hoc, $P <$ 0.001 in all cases). Bullhead was totally absent from S16, upstream of B9, before the connectivity-restoration in Oct 2012. Following the installation of the rock-pool easement, bullhead started to recolonize the upstream reach and they managed to disperse upstream more than 100 m every year. Six years post restoration, they had successfully dispersed to 798 m upstream of the culvert by summer 2018.

For stone loach, the mean densities across the catchment did not vary markedly between sites but significantly declined in S7-S16 from 2017 onwards, with stone loach not being recorded at most of these sites and years (2017-2019), or at negligible densities (Table 5.3; paired post hoc, $P < 0.05$ in all cases). No change in the common minnow population across catchment occurred during the study period (LMM, $F_{6,90} = 0.66$ $P = 0.68$). Between 2013 and 2015, no three-spined stickleback were caught during the electrofishing surveys. Since 2016, three-spined stickleback started colonizing the catchment. The majority of stickleback were caught at S1, within the patch of backwater below the bridge apron, and they are still absent from most upstream sites. Eel were mostly caught in downstream sites (S1-S3) between 2013 and 2015, however the population had a significant increase in 2016 at S1-S5, S7-S9, S11-S13 (Table 5.3; paired post hoc, $P <$ 0.001 in all cases); and they managed to disperse as far as S12. By 2017-2019 eel remained at low densities but were much more widespread than over the period 2013-2015.

5.4.3 Brancepeth Beck

Ten species were caught from Brancepeth Beck during the survey period (note, WRT data from 2015 and 2016 are used in trout demography analyses later in this section; data not shown, available from author or WRT upon request). The predominant species was brown trout (Figure 5.18), which was recorded at majority sites through the catchment. Atlantic salmon was not caught at any site during the study. Bullhead and stone loach were present at a slightly lower abundance than trout. Bullhead were also present through the whole catchment, and occurred at a higher density than trout in the most upstream unrestored sites (Figure 5.19). Stone loach was found widely dispersed through the river in 2014, but was totally absent from all sites located upstream of B4 since 2017 (Figure 5.20, 5.21). Similar to stone loach, common minnow was caught in the majority of sites in 2014. However, this species had totally disappeared between S3 and S25 in 2017, and seemed to be slowly recolonizing upstream in 2019 (Figure 5.20, Figure 5.21). Three-spined stickleback, European eel, brook lamprey and chub (*Squalius cephalus*) were present in very low densities (Figure 5.20-5.21). Most three-spined stickleback were caught in the top and bottom reach of the catchment, this species was totally absent from the mid-catchment. The top of the catchment was characterised by relatively shallow, slower flowing habitat. Brook lamprey were only captured in the most downstream reference sites (S1) and chub were only caught between S1 and S3. One roach (*Rutilus rutilus*) was caught in S1 in 2018 and one perch was caught in S5 in 2018. Both species were excluded from the following analyses due to the low abundance (<0.01% of total fish caught).

Capture efficiencies were calculated for three fish species with differing habits: solitary midwater (trout), solitary benthic cryptic (bullhead) and schooling midwater (minnow). The mean estimated efficiency (all sites combined) of three-pass fishing ranged from 89.2% to 93.93% for YoY trout; 97.4% to 99.7% for older trout; 82.6% to 95.1% for bullhead and 79.1% to 95.7% for minnow in Brancepeth Beck between 2017 and 2019 (Table s5.3).

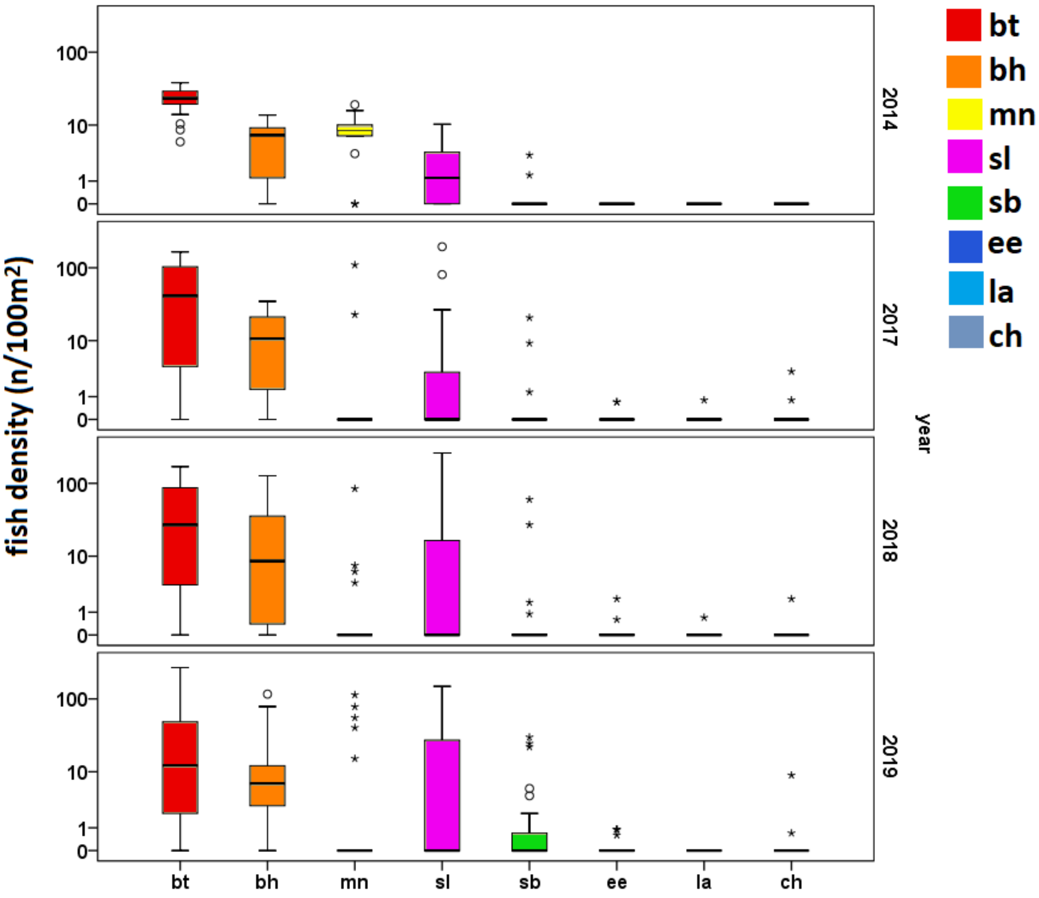


Figure 5.18 Fish density (note log scale) of each species through the sampling periods at Brancepeth Beck (from three pass electro-fishing). BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, LA: brook lamprey, CH: chub. Notice: S1-S3 and S25 were not surveyed in 2014.

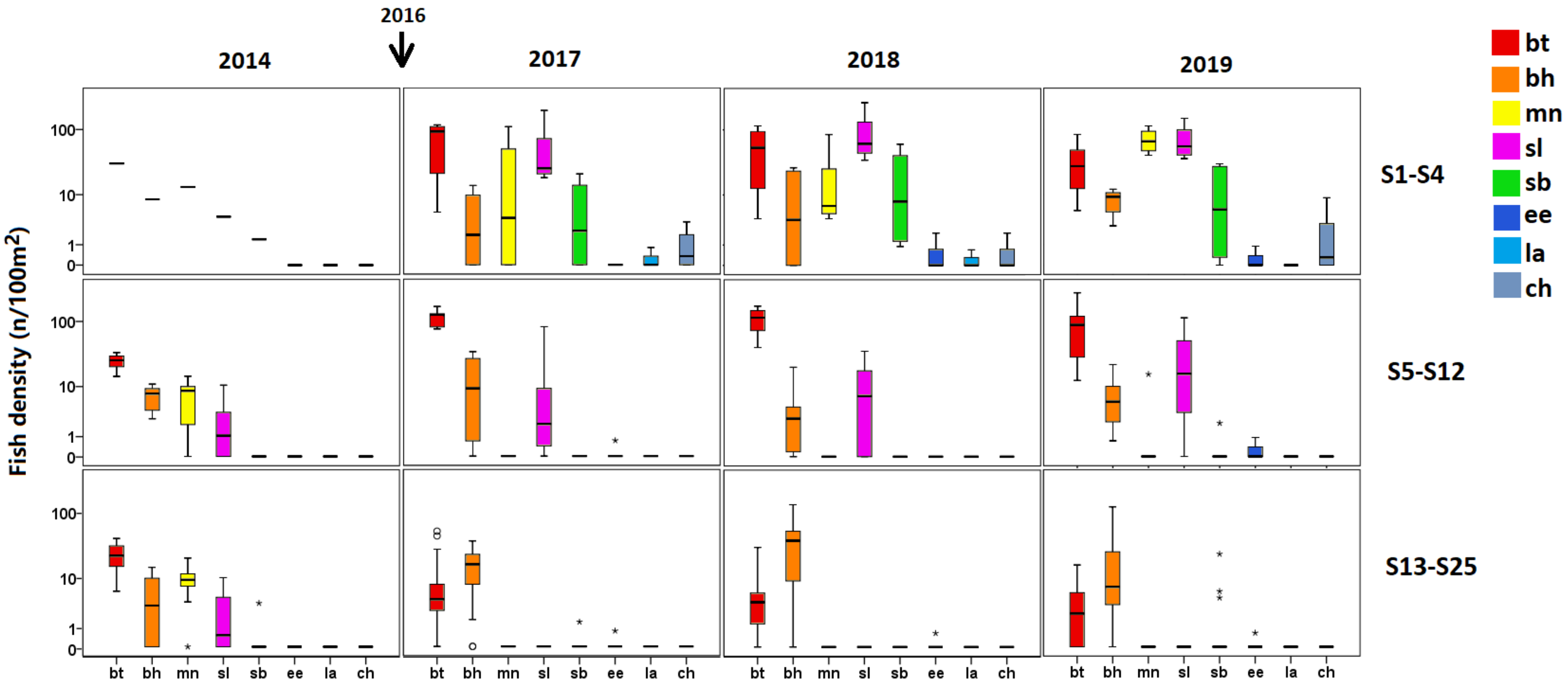


Figure 5.19 Boxplots showing the variation in fish species density (note log scale) between different stream reaches before and after connectivity restoration (shown by arrow, 2016) in Brancepeth Beck. Sampling by three pass electric fishing; S1, S2, S3 and S25 were not surveyed in 2014. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, LA: brook lamprey, CH: chub.

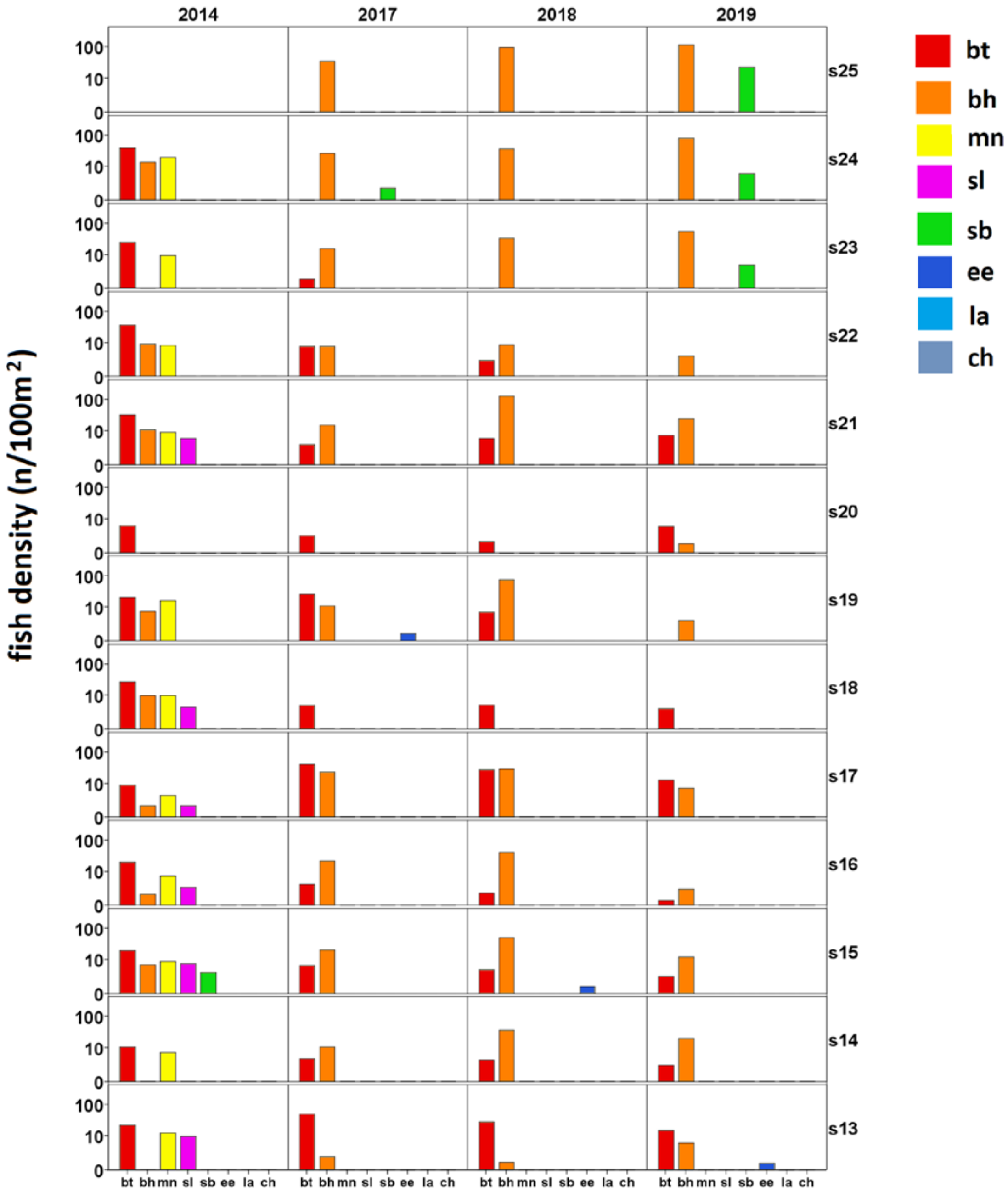


Figure 5.20 Density of fish species in Brancepeth Beck between S13 and S25 across years, sampled by three pass electric fishing. S25 was not surveyed in 2014. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, LA: brook lamprey, CH: chub.

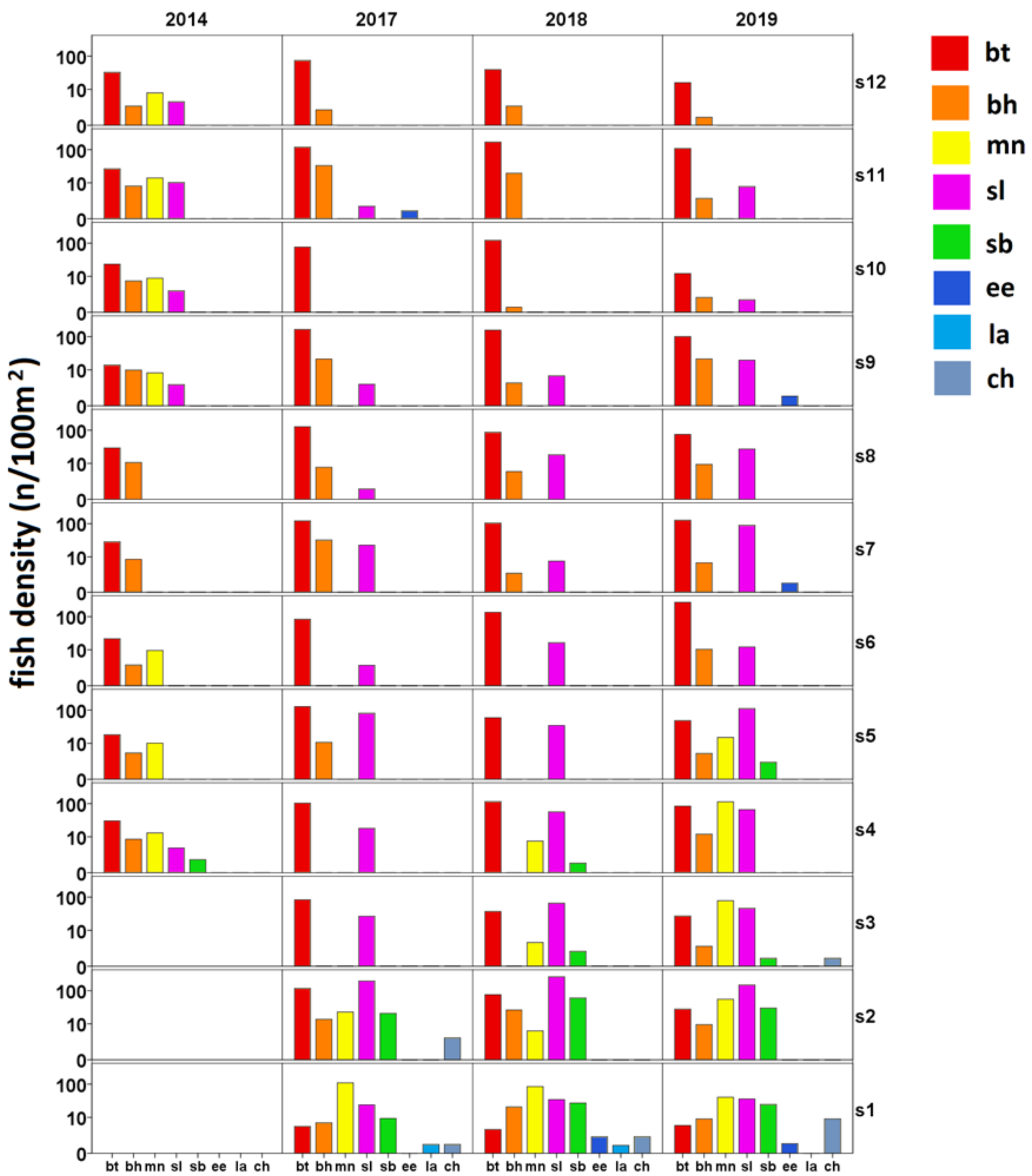


Figure 5.21 Density of fish species in Brancepeth Beck between S1 and S12 across years, sampled by three pass electric fishing. S1, S2, S3 and S25 were not surveyed in 2014. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, LA: brook lamprey, CH: chub.

Post restoration, significant differences in the fish community were found across the catchment (S5-S12, impact with connectivity mitigation of successive barriers; S13-S20, ‘control’ without barrier interventions) between the pre-intervention year (2014) and the post restoration year (2017) (PERMANOVA, $F_{1,28}$ = 4.50, P = 0.023; Table 5.4). No significant differences in fish community were found between the post-intervention years (2017-2019, PERMANOVA, P > 0.05 in both cases; Table 5.4). In addition, no significant differences were found in the fish community between 2016 and 2017 (WRT single pass survey in 2016 vs first fishing in 2017; PERMANOVA, $F_{1,26}$ = 2.73, P = 0.095). Minnow, bullhead, brown trout and stone loach contributed 96.1% dissimilarity in the fish communities in the overall catchment between 2014 and 2017, minnow and brown trout varied significantly post restoration (SIMPER, P < 0.05 in both cases; Table 5.5). A similar situation was observed in the connectivity-restored sites (S4-S12), and the same four species contributed 97.6% dissimilarity in the fish communities between 2014 and 2017. Minnow and brown trout abundance changed significantly post restoration, compared to before (SIMPER, P < 0.05 in both cases; Table 5.5), brown trout contributed 23.7% dissimilarity within the restored reach. Within the unrestored reach (S13-S20), minnow and stone loach changed significantly between 2014 and 2017 (SIMPER, P < 0.05 in both cases). Brown trout contributed 14% dissimilarity within the unrestored reach.

Table 5.4 Upper panel: BACI comparisons of fish communities in the Brancepeth Beck before and after connectivity restoration (2014 vs 2017), Impact: S5-S12; Control: S13-S20. Lower panel: fish communities change during post-intervention years (2017-2019).

Year		***df***	***F***	***P***
2014 vs 2017	BA		11.57	**0.001**
	CI		2.47	0.097
	BA x CI	1,28	4.50	**0.023**
2017 vs 2018		1,30	-0.13	0.952
2018 vs 2019		1,30	1.58	0.213

Table 5.5 Results of the SIMPER analysis (Before vs After [2017 only]), based on Bray-Curtis dissimilarity index, showing status of six fish species (full names given in Figure 5.21 legend, above) response to the connectivity restoration in Brancepeth Beck.

2014 vs 2017	***Species***	***Contribution (%)***	***P***
All site combined (S5-S20)	mn	32.42	**0.001**
	bh	23.15	0.510
	bt	20.52	**0.020**
	sl	19.97	0.266
	ee	2.36	0.626
	sb	1.58	0.120
Restored section (S5-S12)	mn	30.19	**0.007**
	bt	23.7	**0.001**
	sl	23.57	0.552
	bh	20.18	0.087
	ee	2.36	0.603
Unrestored section (S13-S20)	mn	34.93	**0.002**
	bh	26.6	0.488
	sl	19.06	**0.018**
	bt	13.99	0.268
	sb	3.04	0.066
	ee	2.38	0.974

Total fish density was significantly greater in the reconnected reach (in which fish passage provisions were made) than the unrestored reach upstream (Table 5.6; LMM, $F_{1,58}$ = 4.57, P = 0.04). The connectivity restoration time period also had a significant effect on total fish density (LMM, $F_{1,23}$ = 13.03, P = 0.02), with mean total fish density across all connectivity-restored sites nearly doubling after the restoration. A significant interaction effect between site location and restoration status (LMM, $F_{1,58}$ = 20.84, P < 0.01) indicated that fish density changed in response to restoration status. During the study period, the lowest average total fish density (40.9 ± 15.5 per 100 m^2) occurred in 2014; and the highest total density (99.8 ± 86.6 per 100m^2) occurred in 2018. The fact that fish densities in the connectivity-restored reach remained lower than those in the reference reach in 2018 and 2019 suggests that connectivity restoration effects have been partial (incomplete), but positive, in returning the fish community towards reference conditions (Figure 5.22).

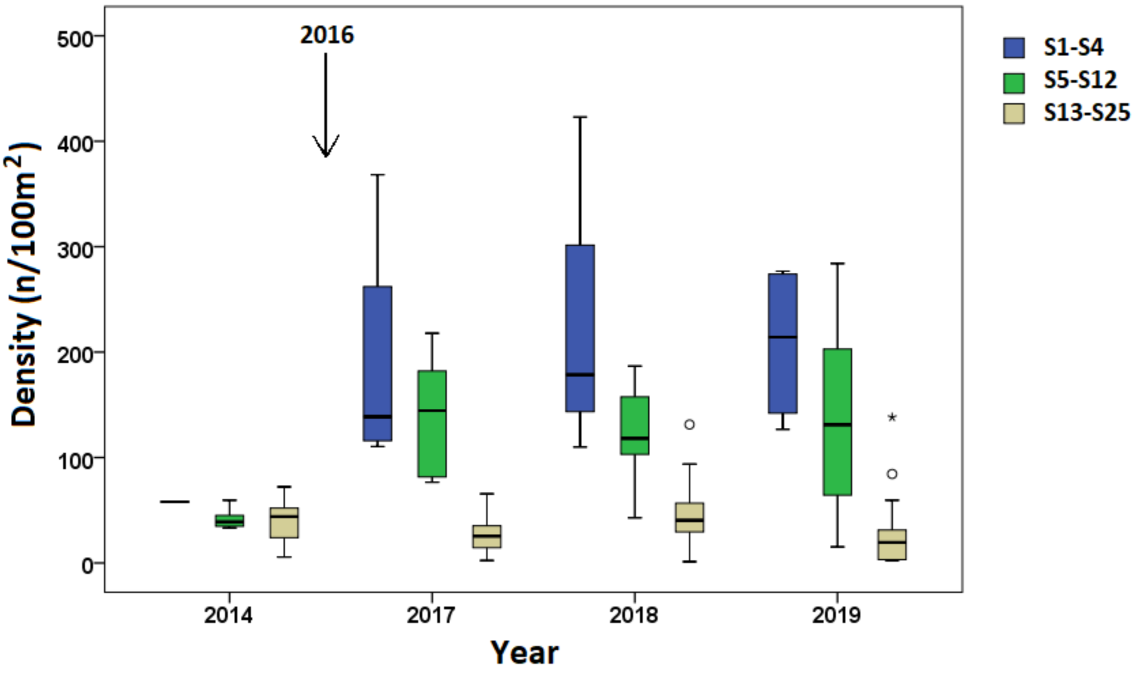


Figure 5.22 The variation in total fish density between reference sites without barriers (S1-S4), the connectivity-restored reach (S5-S12) and the unrestored reach (S13-S25) of Brancepeth Beck across years. Sampling was by three pass electric fishing; S1, S2, S3 and S25 were not surveyed in 2014. The arrow indicates the timing of connectivity restoration.

Significant differences were found in brown trout and stone loach densities between the control (unrestored) and impact (restored) reach (Table 5.6; LMM, brown trout, $F_{1,23}$ = 25.02, P < 0.001; stone loach, $F_{1,27}$ = 13.45, P = 0.001). Mean stone loach density (all years combined pre and post restoration) in the restored reach (15.2 ± 27.2 per 100m^2) was significantly higher compared with the density in the unrestored reach (0.6 ± 1.9 per 100m^2, Figure 5.19). Brown trout also increased in the restored zone (see further information below). Minnow density (2017 - 2019 combined) reduced significantly post restoration by comparison with the density in 2014 (Table 5.6; LMM, $F_{1,58}$ = 201.52, P < 0.001). The mean density was 8.6 ± 4.9 per 100m^2 in 2014, and reduced to zero in 2017 (Figure 5.19). However, no minnow were caught during the WRT fish surveys in both 2015 and 2016 (data not shown, available from author or WRT upon request) within the restored and unrestored reaches, suggesting minnow had already vanished in these reaches of the stream between summer 2014 and summer 2015. Brown trout, bullhead and stone loach exhibited a significant interaction effect between site location and restoration status during the sampling periods (Table 5.6; LMM, *P* < 0.001 in all cases). Across all sites combined mean bullhead density increased from 5.8 ± 4.2 per 100m^2 in 2014 to 12.5 ± 11.0 per 100m^2 in 2017, then remained stable (Figure 5.19). No differences were found in three-spined stickleback and eel density between restored and unrestored reaches, as well as between before and after restoration during the study periods (Table 5.6; LMM, *P* > 0.05 in all cases).

Table 5.6 Comparison of changes in fish densities over before and after connectivity restoration (2014 vs 2017, 2018, 2019) between restored section (S5 - S12) and unrestored section (S13 – S25). BA: before after, CI: control impact, BA:CI: interaction effect.

Species	**Comparison**	***df***	***F***	***P***
Total	BA	1,58	4.57	**0.037**
	CI	1,23	13.03	**0.002**
	BA:CI	1,58	20.84	**<0.001**
Brown trout	BA	1,58	0.01	0.928
	CI	1,23	25.02	**<0.001**
	BA:CI	1,58	41.98	**<0.001**
Bullhead	BA	1,58	1.91	0.172
	CI	1,21	0.05	0.827
	BA:CI	1,58	8.31	**0.006**
Stone loach	BA	1,58	0.00	0.976
	CI	1,27	13.45	**0.001**
	BA:CI	1,58	17.67	**<0.001**
Eel	BA	1,76	2.24	0.139
	CI	1,76	0.12	0.729
	BA:CI	1,76	0.12	0.729
Stickleback	BA	1,58	0.09	0.768
	CI	1,28	0.89	0.354
	BA:CI	1,58	0.09	0.761
Minnow	BA	1,58	201.52	**<0.001**
	CI	1,58	0.84	0.366
	BA:CI	1,58	3.36	0.072

Within the connectivity-restored reach, mean brown trout density increased dramatically from 24.7 ± 5.9 per 100m^2 in 2014 to 111.1 ± 28.5 per 100m^2 in 2017, then remained steady (Figure 5.23). In contrast, within the unrestored reach, the trout density gradually decreased from 22.4 ± 10.3 per 100m^2 in 2014 to 3.7 ± 4.9 per 100m^2 in 2019. The minimum 0+ trout density in the restored reach increased significantly one year after connectivity restoration, from 11.5 ± 10.1 per 100m^2 to 56.6 ± 12.2 per 100m^2 (Table 5.7; 2016 vs 2017; LMM, $F_{1,7}$ = 22, P < 0.005). In contrast, no significant change was found in 0+ trout density within the unrestored reach (2016 vs 2017; LMM, $F_{1,10}$ = 1.24, P = 0.29). In 2017, the 0+ trout density (from autumn 2016 spawning) in the restored reach was significantly higher compared with the unrestored reach (Table 5.7; Figure 5.23; LMM, $F_{1,10}$ = 210, P < 0.001). However, there was no significant difference in the older trout density between the two reaches (Table 5.7; LMM, $F_{1,19}$ = 0.41, P = 0.531). In 2018 and 2019, both 0+ and older trout density were significantly higher in the restored reach comparing with the unrestored reach (Table 5.7; LMM, P ≤ 0.001 in all cases). Trout densities in the restored reach were as high or higher than those in the reference reach (Figure 5.23).

Table 5.7 Upper part: comparison of changes in minimum fish densities from single pass (or first fishing of multi pass surveys) electro-fishing surveys before and after connectivity restoration (2016 vs 2017) between restored section (S5 - S12) and unrestored section (S13 – S20). Lower part : comparison of changes in trout fish densities from three pass electro-fishing survey over restored section (S5 - S12) and unrestored section (S13 – S25) after the connectivity restoration.

Year	**Reach**	**Species**	***df***	***F***	***P***
2016 vs 2017	restored	YoY trout	1,7	22.00	**0.002**
	unrestored	YoY trout	1,10	1.24	0.292
	restored	Total trout	1,7	16.92	**0.004**
	unrestored	Total trout	1,5	0.01	0.914
	restored	Bullhead	1,7	6.36	**0.040**
	unrestored	Bullhead	1,5	9.01	**0.030**
	restored	Stone loach	1,7	9.39	**0.018**
2017	Restored vs Unrestored	YoY trout	1,10	210.00	**<0.001**
		Older trout	1,19	0.41	0.531
2018		YoY trout	1,17	30.11	**<0.001**
		Older trout	1,19	58.72	**<0.001**
2019		YoY trout	1,19	14.70	**0.001**
		Older trout	1,12	16.40	**0.001**

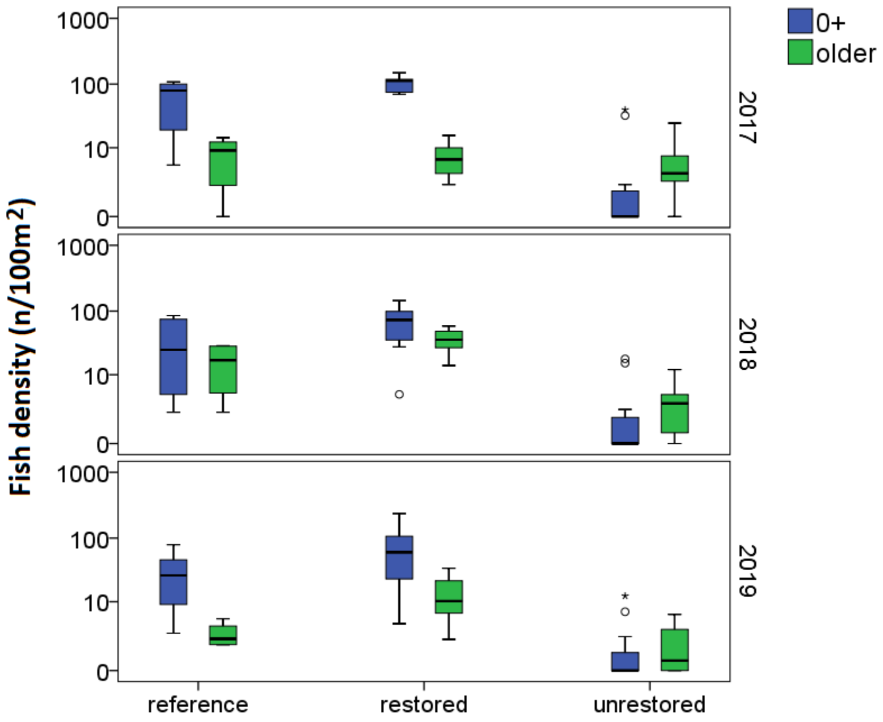


Figure 5.23 Boxplots of 0+ and older brown trout density (from three pass electric fishing surveys, note log scale of abundance) between different stream reaches of Brancepeth Beck after connectivity restoration.

The length of brown trout also differed significantly across the sampling periods from 2016 to 2019 (Figure 5.24; Kruskal-Wallis *H* Test, $X^2(3) = 473.92$, $P < 0.001$) being significantly longer in 2016 compared with post restoration sampling years (Post hoc, $P < 0.001$ in all cases). In 2016, before the effects of restoration, trout length in the upstream unrestored reach was markedly longer compared with the restoration reach (Mann-Whitney *U* Test, $U = 36288$, $P < 0.001$). After connectivity restoration, in 2017, 2018 and 2019, trout length in the upstream unrestored reach remained significantly longer (likely indicative of older fish) compared with the restoration reach and the downstream reference reach (Kruskal-Wallis *H* Test, $X^2(2) = 136.76$, $P < 0.001$).

During anadromous salmonid redd surveys covering the whole stream length in autumn 2017 (13th and 26th November; 4th December), four redds were identified between S1 and S2, and seven redds were recorded between B1 and B5 (Figure 5.25). No redds were found upstream of B5. In addition, an adult sea trout was observed in the reach immediately upstream of B3. During the 2018 redd survey (26th November), one redd was recorded downstream of S1, 15 redds were recorded between S1 and S2, ~six adult sea trout were observed between S1 and S2, seven redds were recorded between S4 and S5, four redds were recorded between B1 and B5. No redds were found further upstream.

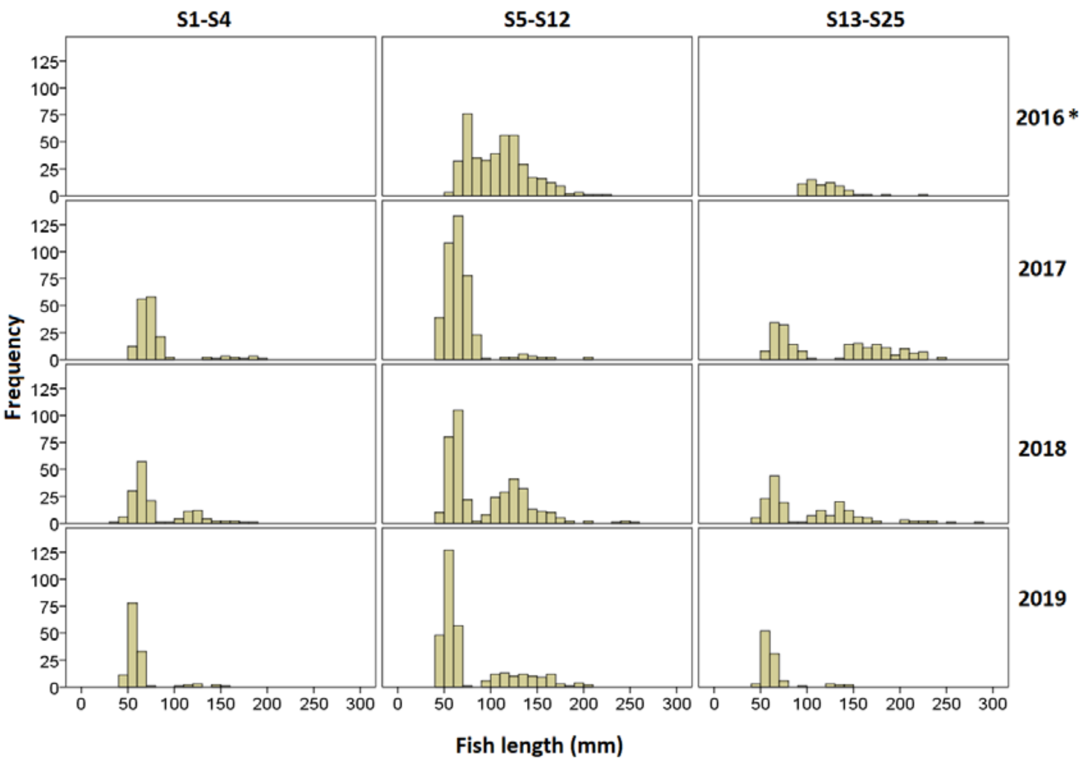


Figure 5.24 The length frequency distribution of brown trout during the study period. The 2016 fish data were extracted from the WRT single-fishing data set (available from the author or WRT). Note: S1-S4, S15, S16, S21-S25 were not surveyed in 2016. The 2015 data were not included here because fish length data were only available at five sites. Trout lengths were not measured in the surveys in 2014 by Tummers (2016).

Figure 5.25 Top and Bottom left: Sea trout redds recorded during surveys of Brancepeth Beck (Photo Credit: Jingrui Sun). Bottom right: dead sea trout found in the beck (Photo Credit: Wear Rivers Trust).

5.4.4 Cong Burn

Seven fish species were captured from the Cong Burn system during the Durham University single pass electro-fishing survey period of 2017-2019 (Figure 5.26, 5.27). The dominant species was brown trout, found at all sites except S14. Other species were all present in lower densities. Common minnow and Atlantic salmon were only caught at S1, bullhead and stone loach were only caught in S1 and S2. These four species were totally absent from sites further upstream.

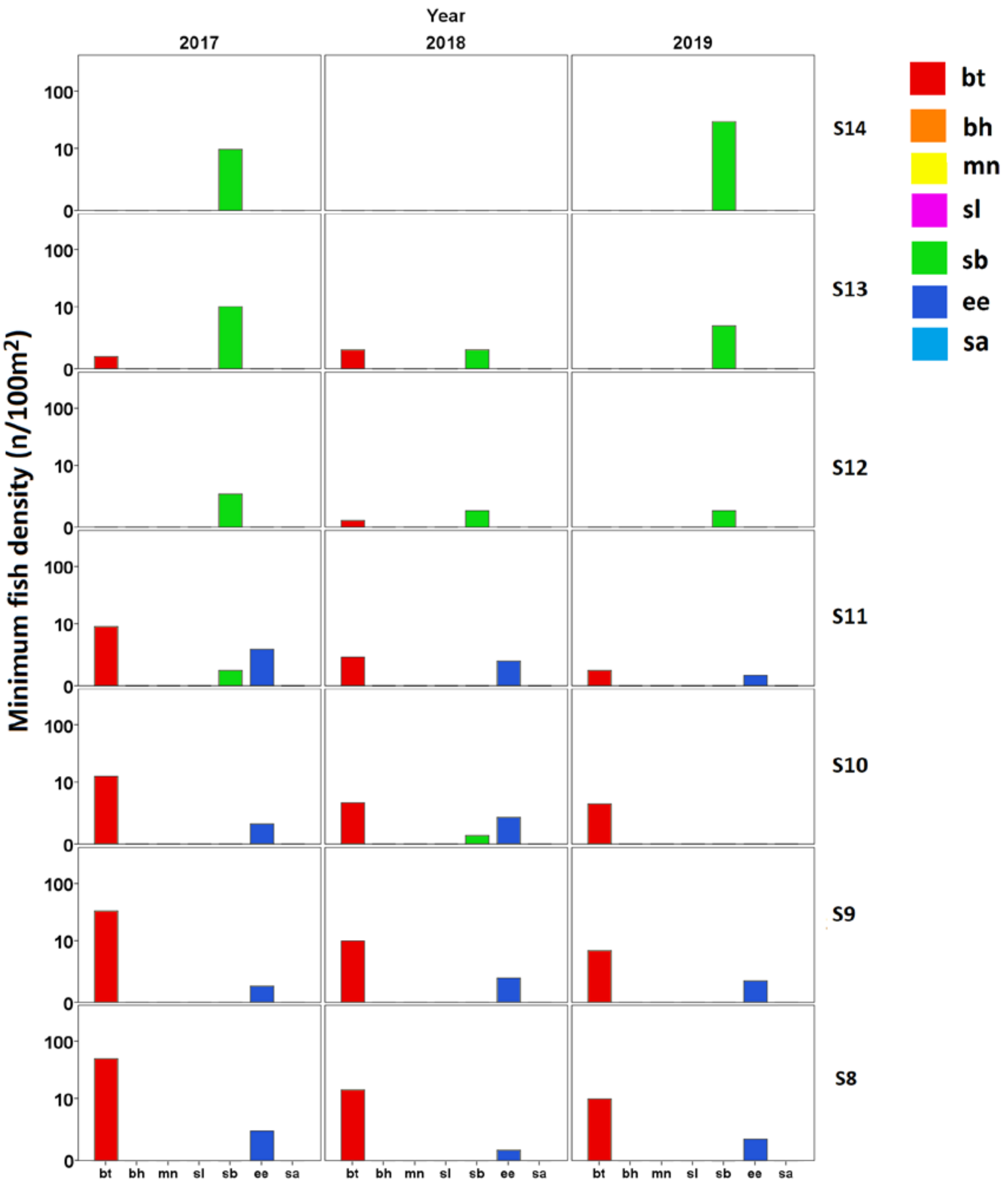


Figure 5.26 Minimum density of fish species (note log scale) between S8 and S14 during the sampling period at Cong Burn. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, SA: Atlantic salmon.

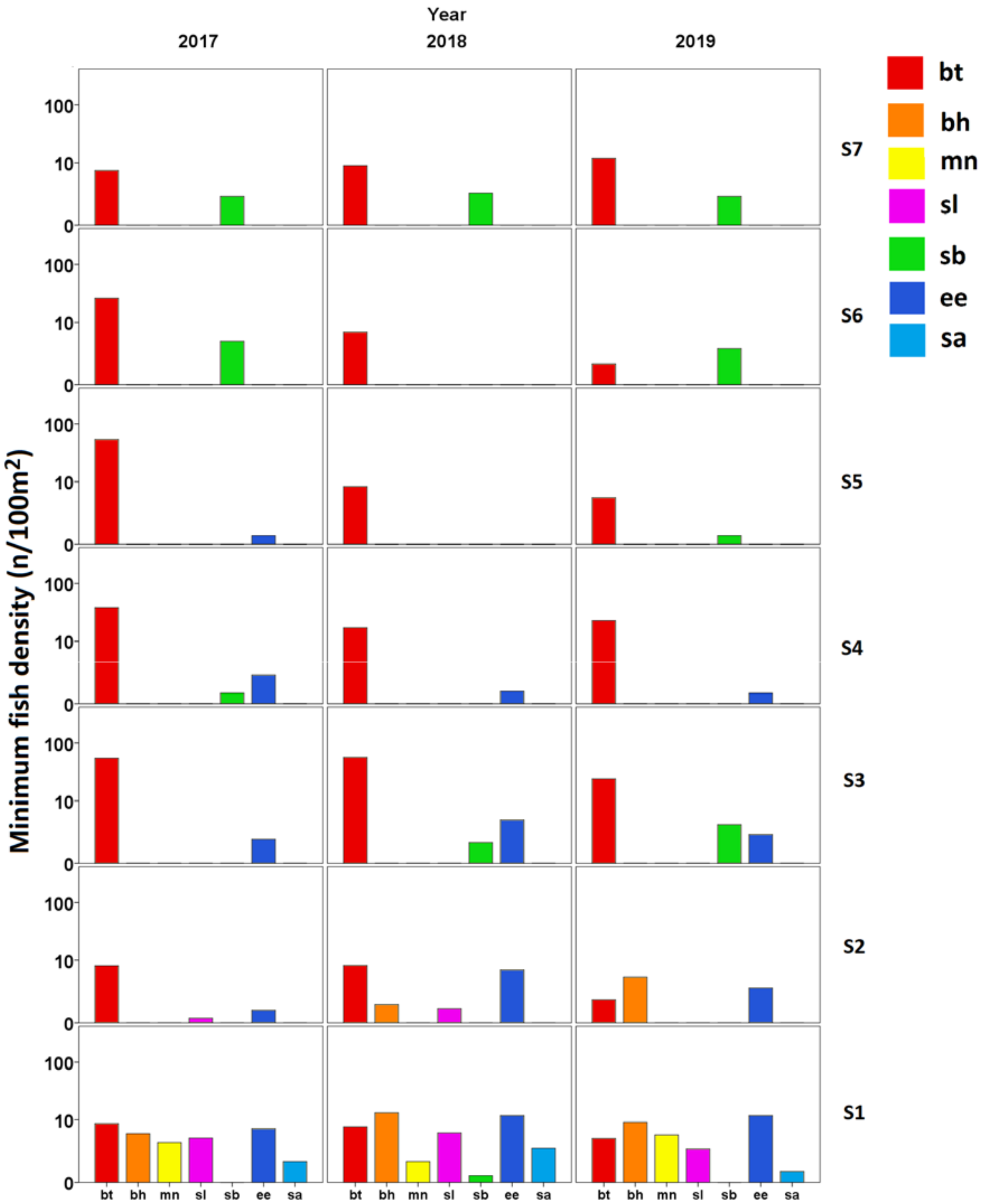


Figure 5.27 Minimum density (single pass electric fishing) of fish species (note log scale) between S1 and S7 during the sampling period at Cong Burn. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, SB: three-spined stickleback, EE: eel, SA: Atlantic salmon.

No significant differences in the fish community were found between 2017, 2018 and 2019 (PERMANOVA, $P > 0.05$ in both cases). Among all fish species, no significant differences in stickleback and eel density were found during the study period (Table 5.8; LMM, $P >$ 0.05 in both cases). No LMM analyses were performed for minnow, salmon, bullhead and stone loach because they were absent from the majority of sites and unequally distributed.

Table 5.8 LMM results (table sections show separate model outputs) showing the change of minimum fish density in the Cong Burn catchment across the study years. BT: brown trout; SB: stickleback; EE: eel. Site was used as a random factor in the analysis.

Species	*df*	*F*	*P*
Total	2,26	5.12	**0.01**
BT	2,26	11.98	**<0.001**
SB	2,26	1.02	0.37
EE	2,26	1.32	0.28
BT 0+	2,26	21.42	**<0.001**
BT older	2,26	20.51	**<0.001**

The brown trout density significantly reduced during the study period (Figure 5.28; LMM, $F_{2,26} = 11.98$, $P < 0.001$). In addition, the 0+ trout density also exhibited a significant decrease through the period (LMM, $F_{2,26} = 21.42$, $P < 0.001$). In 2018, 0+ trout were absent from most sites, the density was significantly lower compared with the previous year (Post hoc, $P < 0.001$). The older trout density showed a significant increase and then reduction through the period (LMM, $F_{2,26} = 20.51$, $P < 0.001$). Among all sampling sites, S3 had the highest trout density in all three years. The long term monitoring data from the EA (Figure 2.32, 2003-2019) and WRT (Figure 5.29) indicate the minimum trout density in the lower Cong Burn, including S3 and S4, showed an increasing trend since 2011, peaked in 2014 and steadily decreased until 2019 (Figure 5.29). A similar pattern was evident at S6, S7 and S10, the minimum trout density from WRT surveys showed an increasing trend between 2013 and 2015, then it decreased steadily until 2019.

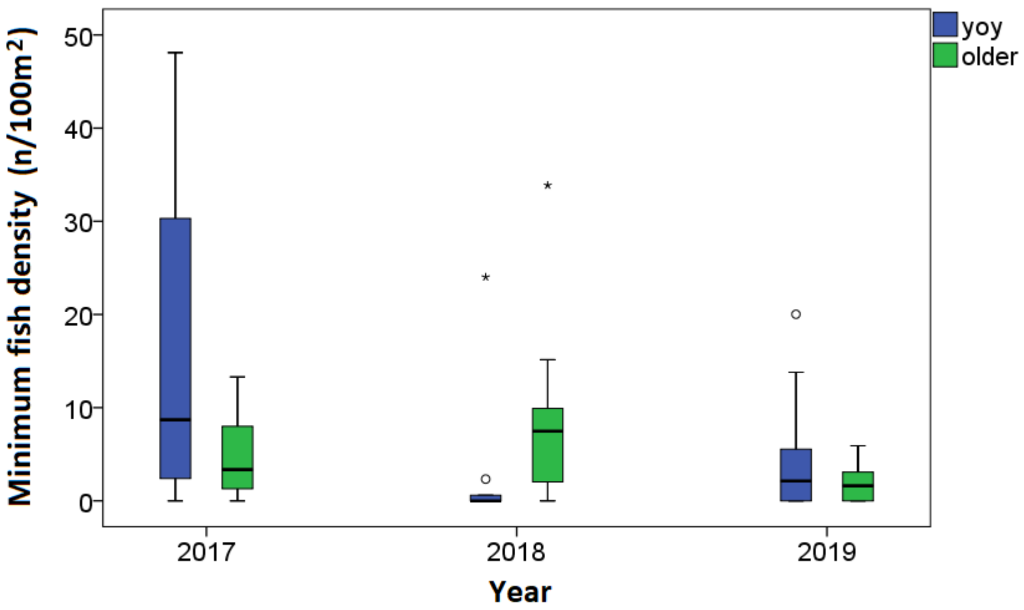


Figure 5.28 Boxplots showing the variation between 0+ and older brown trout minimum density during the sampling periods in Cong Burn.

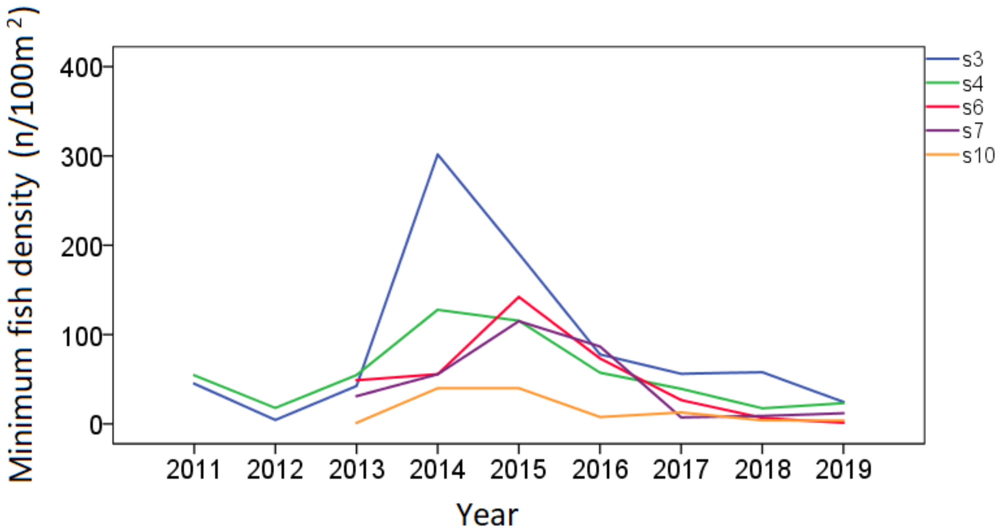


Figure 5.29 The variation of minimum trout density at sites between 2011 and 2019 in the Cong Burn. Data from 2011 to 2016 were extracted from WRT single pass electro-fishing. Data from 2017 to 2019 were extracted from Durham University single pass electro-fishing. S3 and S4 were restored in 2010, S6 and S7 were restored in 2012, and S10 was restored in 2016.

5.4.5 Bedburn Beck

Six species were caught during the study period 2017-2019 (Figure 5.30). Brown trout and bullhead were the predominant species in the river. Atlantic salmon were present at a slightly lower abundance but were recorded at all sites. Stone loach, minnow and eel were only captured at S1 and were absent in both S2 and S3 during the study period. In addition, brook lamprey was captured at S2 during the EA electro-fishing survey periods. The only site sampled by both the EA and Durham Uni was S2 in 2019. The EA fish density (minimum trout 5.3 per 100m^2; salmon 4.3 per 100m^2) is higher than Durham fish density (minimum trout 1.9 per 100m^2; salmon 2.9 per 100m^2).

The first round capture efficiency of EA's three pass electrofishing (2001-2006, 2019) at S2 were calculated for two types of fish species: solitary midwater (trout and salmon). The mean estimated first round efficiency of three-pass fishing is 60.5% for trout and 49.8% for salmon. The capture efficiency of three pass electrofishing can be found at Figure 2.35.

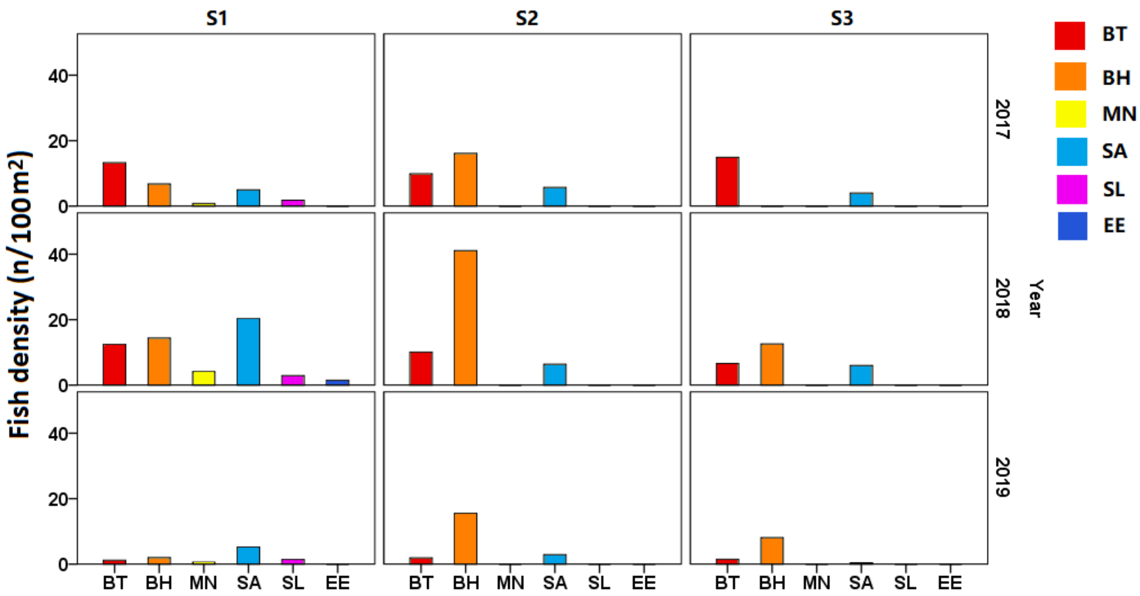


Figure 5.30 Minimum density of fish species (from single pass electric fishing) between S1 and S3 during the sampling periods at Bedburn Beck. BT: brown trout, BH: bullhead, MN: minnow, SL: stone loach, EE: eel, SA: Atlantic salmon.

For trout, both 0+ and older trout minimum density showed a steadily decreasing trend between 2017 and 2019 within the three study sites (Figure 5.31). For salmon, both 0+ and older salmon minimum density slightly increased in 2018, then sharply reduced in

2019. Combining with EA data, the long-term YoY trout minimum densities in S2 were stable between 14.9 and 61.0 per 100 m^2 between 1991 and 2006, then largely decreased below 20 per 100 m^2 in most samplings since 2007 (Figure 5.32). A linear regression established that YoY trout density statistically significantly declined across years ($F_{1, 18}$ = 13.15, R^2 = 0.422, P = 0.002). The long-term YoY salmon minimum densities varied between 2.3 and 35.0 per 100 m^2 from 1991 to 2014, then largely reduced below five per 100 m^2 since 2016. A linear regression established that YoY salmon density statistically significantly declined across years ($F_{1, 18}$ = 13.1, R^2 = 0.421, P = 0.002). No statistically significant trend was found between older salmon / trout densities and years (Trout: $F_{1, 17}$ = 0.455, R^2 = 0.026, P = 0.509; salmon: $F_{1, 18}$ = 1.187, R^2 = 0.062, P = 0.29). In S2, the overall salmon and trout minimum density varied considerably between 1991 and 2010 (Figure 5.33). Since 2011, both salmon and trout density steadily decreased and reached their lowest value in 2019. By contrast, the bullhead was absent or present in a very low density in S2 between 1995 and 2010, then increased dramatically since 2011 and became steady until 2019 (Figure 5.32). A moderate negative correlation was found between bullhead and trout density from 1995 to 2019 (Spearman's Rank Correlation, r = -0.49, P = 0.03). A moderate negative correlation was found between bullhead and salmon density from 1995 to 2019 (Spearman's Rank Correlation, r = -0.52, P = 0.02). A strong positive correlation was found between trout and salmon density from 1991 to 2019 (Spearman's Rank Correlation, r = 0.77, P < 0.001).

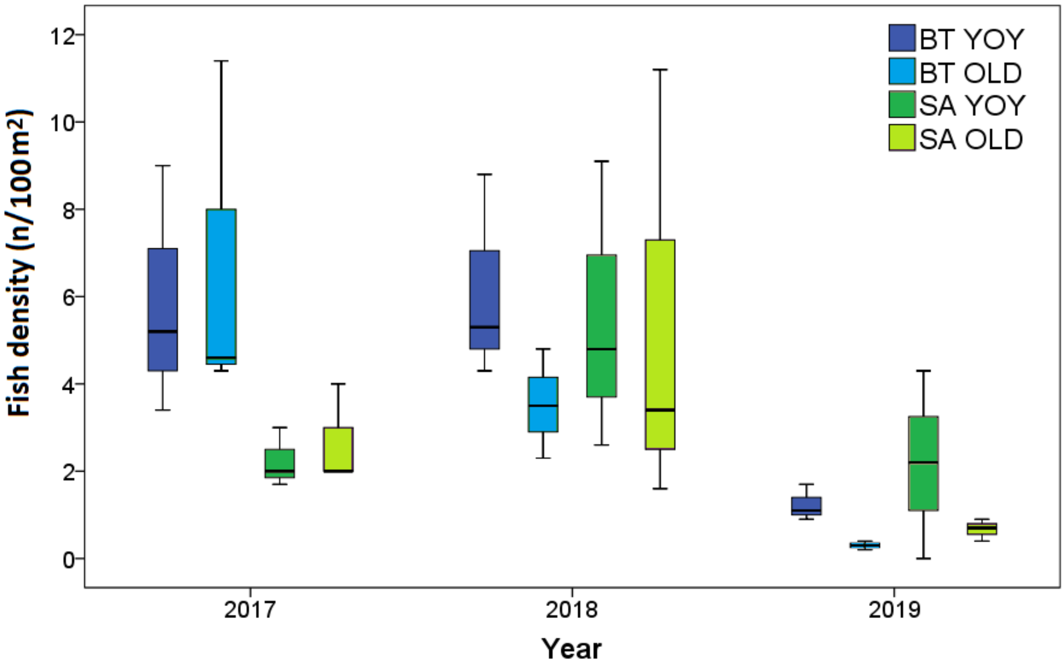


Figure 5.31 Boxplots showing the variation between 0+ and older brown trout and Atlantic salmon (single fishing, minimum density) during the sampling periods.

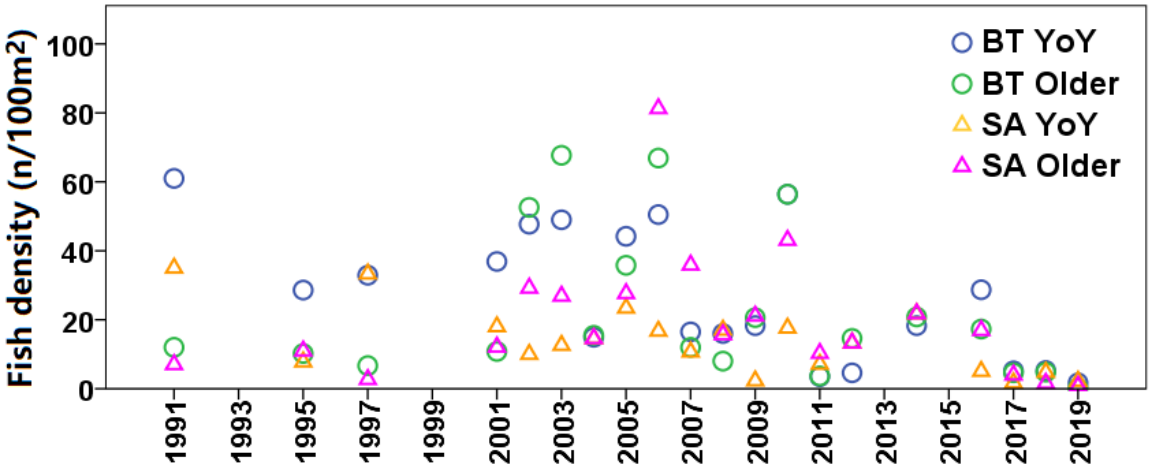


Figure 5.32 Long-term variation of minimum trout and salmon density in S2 at Bedburn Beck. Data between 1991 and 2019 were extracted from EA fishing database. BT: brown trout, SA: Atlantic salmon. Note: blank means no survey was conducted in that year.

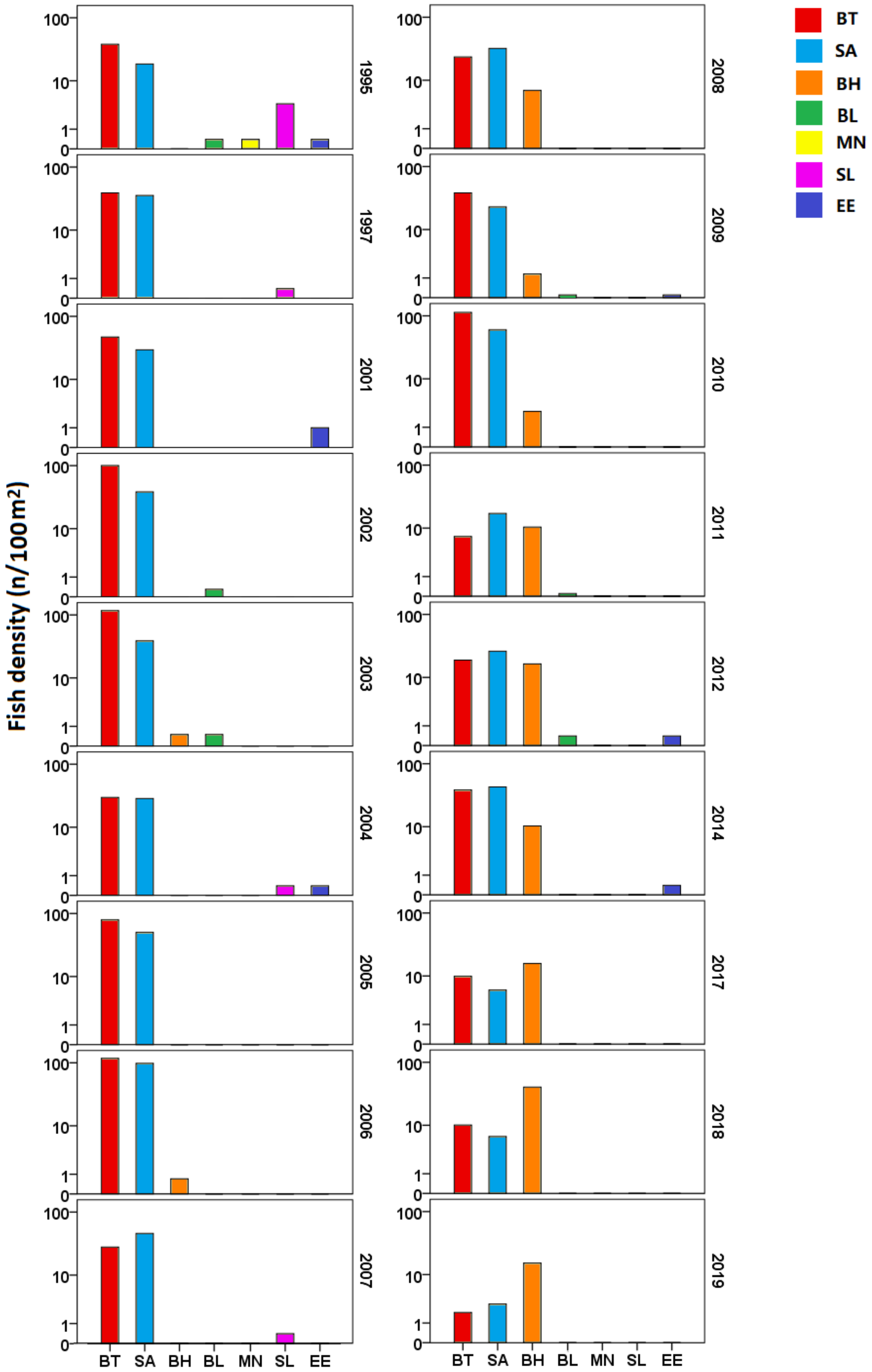


Figure 5.33 Long-term variation of minimum fish density in S2 at Bedburn Beck. Data between 1995 and 2014 were extracted from EA fishing database. BT: brown trout, SA: Atlantic salmon, BH: bullhead, BL: brook lamprey, MN: minnow, SL: stone loach, EE: eel.

A moderate positive correlation was found between minimum YoY trout density and autumn high flow event days (number of days with flows exceeding Q10 and Q5, from 1 September to 15 December) in the previous autumn from 1991 to 2019 (Figure 5.34; Spearman's Rank Correlation, r = 0.46, *P* = 0.04 in both cases). No correlation was found between the minimum YoY trout density and autumn high flow event days (exceeding Q20) as well as the mean autumn flow (mean daily flow from 1 September to 15 December) in the previous year (Spearman's Rank Correlation, *P* > 0.05 in both cases). No correlation was found between minimum YoY trout density and the number of days with flows exceeding Q1 or Q5 between 16 December - 15 May (Figure s5.5; Spearman's Rank Correlation, *P* > 0.05 in both cases).

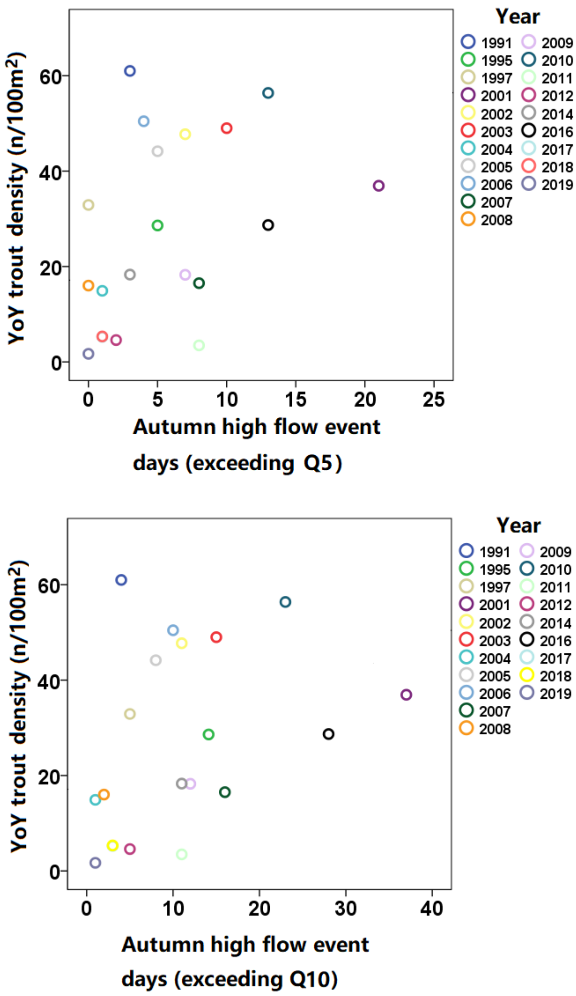


Figure 5.34 Relationship of YoY trout minimum density and high flow event days (exceeding Q5 and Q10) in Bedburn Beck between 1991 and 2019.

No correlation was found between minimum YoY salmon density and mean autumn flow (from 1 September to 15 December in previous year) from 1991 to 2019 as well as autumn/winter high flow days (exceeding Q5 or Q10 or Q20) (Figure s5.6; $P > 0.05$ in all cases). In addition, no correlations were found between minimum YoY salmon density and winter/spring high flow days from 1991 to 2019 (exceeding Q1 or Q5) ($P > 0.05$ in both cases).

Furthermore, no correlation was found between Framwellgate annual fish counter data and mean flow at Chester-Le-Street gauge station from 1 June to 30 November, as well as number of high flow event days (exceeding Q5 or Q10 or Q20) during the same periods (Figure s5.7; $P > 0.05$ in all cases).

5.5 Discussion

The study's key aims, regarding subcatchment-scale connectivity restoration on native fish communities were met. Firstly, the study supports the view that, to achieve a relatively full recovery of the fish community over a whole subcatchment, a strategy must be adopted to solve all sources of pressure. Not only must good connectivity be achieved throughout the subcatchment, but pollution sources need to be removed, and instream and riparian habitat need to be appropriate. Incomplete recovery of water quality, combined with extensive, poor-quality urban habitat, and very incomplete connectivity restoration, seems to have precluded substantive recovery of the natural fish community through most of the Cong Burn, by comparison to the other two studied 'restored' subcatchments. Secondly, the study showed that restoring connectivity at a single site within a highly fragmented river system may have very limited beneficial effects on fish diversity and abundance. In Brancepeth Beck, two barriers located in the middle reach had fish passage easements applied, but the fish abundance and diversity did not improve during the study period because the further downstream reach was still fragmented by other barriers. Thirdly, catchment-scale connectivity restoration can greatly benefit the density and distribution of both migratory and river resident fish species. However, such improvements may take several years to develop. In the River Deerness, both brown trout and bullhead significantly increased in abundance three years after the connectivity restoration and remained elevated until 2019, although less changes were found in fish species diversity. Fourthly, compared with fish pass / easement installation, barrier removal is more effective in restoring rheophilic fish abundance immediately upstream through increasing fluvial habitat, and facilitating the movement of poorly dispersing species such as bullhead. In the River Deerness, YoY trout were present in low abundance at sites where a fish pass was installed but which retained ponding upstream of the barrier. In contrast, YoY trout were abundant at sites where the barrier was fully removed.

In order to obtain a better perspective of the degree to which barrier removals and passage mitigations, alongside longer-term water quality improvements, have enabled the restoration of more natural fish communities and species abundance in River Wear subcatchments, it is appropriate to gauge these relative to reference conditions (Palmer *et al.*, 2005). For the streams in question, the reference condition should refer to the undisturbed state before large-scale industrial disturbances (see Chapter 2). However, when pristine conditions no longer exist, reference communities might broadly be indicated by the species in the lowest section close to the confluence or from a nearby free-flowing stream (Woolsey *et al.*, 2007; Mims and Olden, 2013), while recognising that habitat conditions, not least stream width and correlated habitat and biotic factors change with distance from source (Palmer, 2009). In this study, the communities of all four streams were most diverse furthest downstream. Although Bedburn Beck was intended as a reference good-condition stream, it appears that the fish community changed in composition over time (though based only on one long-term study site). The clearest difference in community composition between the reference stream and the restored streams was the consistent presence of Atlantic salmon in the former, but the near absence in the latter. However, eel were much more abundant in Cong Burn than in Bedburn Beck, probably reflecting an upstream-reducing gradient of eel dispersal, given that Cong Burn is downstream in the catchment and the Bedburn Beck is upstream. With regard to assessing responses in abundance of fishes, a better perspective might be obtained from considering these against reference conditions from a much larger sample size of sites, particularly for key indicator species such as brown trout and Atlantic salmon. Such reference assessments of expected abundance of salmonid indicator species in good habitat conditions have been developed across Western Europe, often by classifying into percentiles of abundance, sometimes referenced against correlated habitat criteria (Karr, 1981; Romakkaniemi *et al.*, 2003; Aprahamian *et al.*, 2006; Forseth *et al.*, 2013).

The Environment Agency Fisheries Classification Scheme (EA-FCS) can be applied in the River Deerness, Brancepeth Beck and Bedburn Beck to gauge the degree to which salmonid abundance approximates to expected reference conditions. The EA-FCS was developed by the National Rivers Authority in 1990, to allow comparison of YoY and older salmonid monitoring data with a database derived from percentile categorization of densities from around 1000 survey sites in England and Wales (Mainstone *et al.*, 1994; Taylor, 2017). The classification of salmonid populations is based on a grading scale (A–F) and provides an indication of the status of salmonid populations in study rivers. The EA-FCS grading scheme is translated as follows: Grade A (excellent), Grade B (good), Grade C (fair or average), Grade D (fair/poor), Grade E (poor) and Grade F (fishless). The population density grades for the EA-FCS are detailed in Table 5.9 This grading system allows for comparison to national trends and puts densities into a context of how good or bad they are, and is included in discussion of the recovery of streams below.

Table 5.9 Brown trout and salmon abundance (number per 100 m^2) classifications from Environment Agency Fisheries Classification Scheme (EA-FCS). 1++ refers to fish of one year and older, while 0+ are young of the year.

Species group	A	B	C	D	E	F
0+ bt	≥38.00	17.00-37.99	8.00-16.99	3.00-7.99	0.00-2.99	0.00
1++ bt	≥21.00	12.00-20.99	5.00-11.99	2.00-4.99	0.00-1.99	0.00
0+ sa	≥86.00	45.00-85.99	23.00-45.00	9.00-23.00	0.00-9.00	0.00
1++ sa	≥19.00	10.00-19.00	5.00-10.00	3.00-5.00	0.00-3.00	0.00

5.5.1 River Deerness

The river connectivity restoration between 2012 and 2014 in the River Deerness significantly increased the overall fish abundance in 2017, 2018 and 2019. Both brown trout and bullhead populations were benefitted from the restoration. Unlike the quick recovery of brown trout after a small dam removal in a Danish river (Birnie-Gauvin *et al*., 2017b), the trout population in the Deerness took three years to exhibit a marked increase. This may because the generation time of brown trout is a minimum of ~2 years, and none of the first four downstream barriers was fully removed, so the presence of these barriers may still affect the penetration of spawners and resultant egg deposition to some degree.

Between 2013 and 2016, according to the EA-FCS grading system the majority of sampling sites in the River Deerness were classified as Grade C or D for both YoY and older trout, with few sites classified as Grade B (Table 5.10). Since 2017, following the increased abundance of both YoY and older trout, nearly half of the sampling sites were classified as Grade A and B annually. In the barrier removed reach, S7-S12 were classified as Grade C and D for YoY trout between 2013 and 2015. In 2016, two of those sites improved to Grade B. No sites between S7 and S12 were classified as Grade D since then. From 2017 to 2019, most of the sites where barriers were removed have improved to Grade A or B.

Table 5.10 Brown trout classifications of River Deerness sampling sites under the EA-FCS abundance grading scheme

Year	Species group	A	B	C	D	E	F
2013	YoY		2	8	6		
	Older		3	13			
2014	YoY		2	9	5		
	Older		4	12			
2015	YoY		2	6	8		
	Older		5	5	6		
2016	YoY		5	10	1		
	Older		1	10	5		
2017	YoY	4	6	2	2	2	
	Older	5	3	5	3		
2018	YoY	5	3	4	3	1	
	Older	7	3	5	1		
2019	YoY	1	8	4	2		1
	Older	2	5	7	1	1	

Strongly increased YoY trout abundance across the barrier-removed reach suggests barrier removal successfully restored the river connectivity and improved access to and availability of high quality rearing habitat for trout fry and parr. In addition, the length-frequency distribution of trout shifted from four clear length modes (putatively Age 0+, 1+, 2+ and 3+) in 2013 to two length modes (Age 0+ and 1+) in 2016, and the median fish length progressively reduced through the study period after the restoration, reflecting an increasing proportion of Age 0+ trout. It is very likely this reflects an increase in the proportion of the Deerness trout population (principally at Age 2) emigrating to the main river or to sea. A substantial proportion of Deerness trout spawners are sea trout (Tummers *et al*., 2016). A similar pattern in trout length response to connectivity restoration was observed in River Villestrup, Denmark. After six weirs were removed from the river, they observed a decrease of the average smolt size through the years (Birnie-Gauvin *et al.*, 2018). The changes in length structure suggest the connectivity restoration decreased the population of resident trout in the River Deerness. The migratory behaviour of salmonids is under partial genetic control and has a high heritability (Ferguson, 2006). However, in brown trout there is a strong environmental component influencing migration tendency (Ferguson *et al.*, 2019). This type of strong genetic control is well developed in resident salmonid populations living in habitats where emigration could be disadvantageous (i.e. prohibited by barriers) (Northcote, 1992; Ferguson, 2006). Followed by connectivity restoration, migration of trout became less restricted, the upstream migration of adult sea trout and downstream emigration of trout smolts became easier without getting blocked by barriers, which has led to the genetic control to be less pronounced (Ferguson *et al.*, 2019).

In addition, the large increase in juvenile trout abundance since 2017 may potentially have increased the intraspecific competition for food and habitat (Jonsson and Jonsson, 2006). The availability of food strongly influences the growth of brown trout (Elliott, 1976, 1982), as well as the tendency for trout parr to smolt or remain resident in the river (Morgan and Paveley, 1993; Olsson *et al.*, 2006; Ferguson *et al.*, 2019). Increasing the quantity of food available to the progeny of sea trout has been shown to lead to increased freshwater growth rates and reduce the tendency for juvenile sea trout to smolt, effectively producing brown trout from a sea trout stock (Morgan and Paveley, 1993; Jonsson and Jonsson, 2006). On the contrary, when food levels were low, trout became migrants (Jonsson and Jonsson, 2006; Olsson *et al.*, 2006), migration to sea could increase the feeding opportunities, then lead to increased fecundity (Ferguson, 2006; Thornton, 2008). So it is suggested that these combined factors have led to the progressive reduction of resident brown trout abundance but increased the migratory sea trout abundance in the Deerness.

In 2016, 2017 and 2018, the study area suffered relatively dry weather conditions during the trout prespawning and spawning seasons (between September and December) compared with 2013-2015. It was expected that low autumn flows could result in low spawner access and egg deposition and resultant low fry densities the following year. However, both YoY and older trout abundance still largely increased in 2017-2019. Although data were limited there was no correlation between the total / YoY trout density and mean daily flow in the previous autumn (and an apparent negative correlation between high flow events exceeding Q10 and trout density the following summer). This suggests, for the range of flows observed and limited data adult sea trout could still access upstream spawning habitat during relatively dry autumns and that spawnings were successful.

For sites with fish pass construction, although this has restored fish passage, it does not restore habitat immediately upstream. So there is no benefit for the YoY trout in the reach immediately upstream (e.g. S2, S4 and S16). The sites S4 and S16 were degraded to Grade E due to low YoY trout density in 2017, and this situation continued in both 2018 and 2019. In 2019, no YoY was caught at S4, which was the first time that YoY trout absent from a sampling site. S2 was classified as Grade C and D for YoY trout during the whole study period except in 2018 it was classified as Grade B when the overall trout abundance in Deerness was high.

These results indicate that barrier removal is a more effective way in restoring juvenile trout abundance in the upstream section compared with fish pass construction. Building the fish pass only mitigates the disconnection of fish passage but it is unable to restore the habitat immediately upstream of the barrier back to its natural status. However, the spatial extent of impounded habitat impact on the Deerness and other moderate gradient streams is relatively less than low-gradient streams such as Southern English and Danish trout streams (Birnie-Gauvin et al., 2017a). Adult brown trout prefer to spawn in the gravel and pebble area within the riffle and glide transition zone which contains sufficient flow and well oxygenated water rather than in the impounded slow flowing deep glide (J. Sun, pers. obs.) (Reiser and Wesche, 1977; Shirvell and Dungey, 1983; Armstrong *et al.*, 2003). After emergence, trout fry inhabit shallow riffles within a few hundred meters downstream of their hatched area (Armstrong *et al.,* 2003). Brown trout juveniles < 7 cm, often occupy shallow and slow-flowing areas (< 20-30 cm depth) in the stream, and they will move to deeper areas as they grow (Armstrong *et al.*, 2003).

In the impounded reaches, movement of fine sediments (< 2 mm in size) was blocked by the barriers and infiltrated into the gaps of larger substrates such as gravel, reducing the permeability and reducing the oxygen supply to developing ova (Soulsby *et al.*, 2001; Louhi *et al.*, 2008). High concentrations of fine sediments would result in lower survival rates of the eggs and inhibit the emergence of trout fry. In S2 and S4, the river habitat are both dominant by slow flowing deep glides (> 1 m deep), and the barriers trapped large amounts of fine sand immediately upstream, suitably only for large parr and resident adult trout. So YoY trout were still present in low abundance six years after the restoration, albeit these were local effects. In contrast, the river habitat in S8, S10 and S12 was of a natural status with multiple flow features after the barrier removal. The habitat in these three sites provided high-quality spawning grounds for adult trout and rearing habitat for both 0+ and older trout.

By contrast to brown trout, salmon have still not populated the Deerness to any significant extent, although very low densities occur near the confluence with the Browney and in the Browney itself (Chapter 2, Figure 2.33). This is potentially due to philopatric effects, since salmon and trout would return to their natal rivers, and often tributaries, to breed (Jonsson and Jonsson, 2011). If the stream itself does not hold any salmon population (or has lost its salmon population) and no salmon parr have been restored in it before, it is unlikely that adult salmon will select that stream to breed. In addition, salmon often prefer to spawn in the main river and larger tributaries, while trout often prefer to spawn in smaller headwater streams (Crisp, 2000; Louhi *et al.*, 2008; Jonsson and Jonsson, 2011), so the Deerness may be unfavourable for salmon to spawn. Equally, there are many instances in which salmon spawn in streams the size of the Deerness, within the Wear (Bollihope Burn, Waskerly Beck) and more widely. The loss of salmon from the Deerness and the slow tendency for its recolonization through philopatry could form part of a hysteresis effect (effects that persist after the initial causes giving rise to the effects are removed – here these main effects are water quality, barriers and physical habitat quality). Equally since the (assumed) pre-industrial era salmon population in the Deerness was likely extirpated due to pollution and urbanisation, the ecological recovery may have followed a different path, such that a different fish community may be present and/or habitat conditions are different to those prior to the Industrial Revolution. In ecology, hysteresis is often used to refer to switches in communities or ecosystems between alternative stable states (Beisner *et al.*, 2003), though that may not be the most relevant context here. Because there is no fish community data on the Deerness prior to the Industrial Revolution (indeed we cannot be 100% sure salmon populated the Deerness) so it is not possible to verify the theory.

The bullhead density increased steadily during the study period. Like trout, bullhead rely on fast flowing streams with little sand or silt, but they utilise larger substrate and spawn under cobbles and boulders (Knaepkens *et al.*, 2002). Removal of barriers and reinstatement of high-quality fluvial habitat is expected to benefit spawning, growth and survival of bullhead (Utzinger *et al.*, 2008). However, several studies have shown no relationship between water depth and bullhead density (Tomlinson and Perrow, 2003; Utzinger *et al.*, 2008) and large substrate (cobble, boulder) still persisted in areas upstream of barriers. So, even in S2 and S4, their population still increased during the study periods. Due to their poor swimming and lack of jumping ability, bullhead are unable to pass any in-stream barriers higher than 18-20 cm (Utzinger *et al.*, 2008), so none of the bullhead managed to reach S14 during the study periods. In order to restore the population of this kind of poor swimming ability fish, any barriers with a height of 20 cm or above are suggested to be removed.

Opposite to brown trout and bullhead, the stone loach population strongly decreased in S7-S16 since 2017. Stone loach are known to be more tolerant of sand-silt habitat than salmonids and bullhead (Roussel and Bardonnet, 1997), so probably from ponded areas upstream of barriers. Stone loach may also have suffered from interspecific competition between trout, bullhead and stone loach in the restored stream. A recent study in France found stone loach exhibited narrow trophic niches and high overlaps (> 70% in June) with 0+ salmon parr in a large European river (Floury *et al.*, 2019). In addition, an examination of feeding niche overlap between bullhead and stone loach showed a substantial overlap in the range of prey items utilised by both fish (Mcleish, 2017). Considering the Deerness is a small scale tributary, the prey resources and spatial or temporal habitat would be limited, and interspecific competition might be more intensive between these species. Three-spined stickleback has been found in the study sites since 2016. It is suggested that the stickleback population could originate from small ponds and ditches during high flow events.

This case study demonstrates the extent to which catchment-scale connectivity restoration can affect the density and distribution of both migratory and river resident fish species. In this case, the overall fish population did not respond to the connectivity restoration immediately. It tooks four years to develop, before the rapid increase in the overall fish population happened and it is unclear as to whether a new equilibrium has been reached yet. Certainly eel densities remain low, salmon are absent and brook lamprey are absent, suggesting that full recovery is still some way off. Furthermore, results of this study revealed that barrier removal is more effective in restoring fish abundance in the immediately upstream reach, and facilitating movement of poorly dispersing species, compared with fish pass construction. Findings of this study suggest that any low-head barriers including smaller weirs and bridge aprons should be considered in catchment-scale restoration plans, and aim for barrier removal as the first approach rather than attempt to install an artificial fish pass on the barrier. Crucially, the findings support the idea that effective stream connectivity restoration for fluvial fishes requires connectivity across most barriers within a subcatchment to generate substantive benefit. These findings have important implications for the in-stream barrier management and river restoration works across the world.

Future restoration work in the River Deerness needs to focus on: (1) Restoring the connectivity in the lower part of the Priest Burn tributary. A 1.1 m high culvert apron with three steps and a 0.9 m high wooden weir were identified within a few hundred meters upstream of the Priest Burn / Deerness confluence during the barrier survey (see Chapter 3). (2) Restore the connectivity at Red Burn tributary (see Chapter 3, Figure 3.6). A total of five in-stream barriers were recorded during the barrier survey on Red Burn, including pipe culverts, woody debris within a culvert, a stepped weir and a spillway (see Chapter 3). These barriers were not been recorded in the original restoration scheme (2012-2014), which could prevent / delay the upstream / downstream migration of all species. (3) Mitigate / remediate point source and diffuse pollution in the sub-catchment. Point source pollution from Esh Winning STW is recorded in the Waterbody Action Plan as is the sewage discharge at downstream of Ushaw Moor Bridge. Both locations failed to meet phosphate limits during the EA assessments between 2013 and 2016, and under WFD this is expected to be improved by 2027. To date, physical and riparian habitat in the Deerness is relatively good and diverse and is one of the reasons why connectivity restoration seems to have been effective in improving fluvial fish densities and facilitating more natural fish community structure.

5.5.2 Brancepeth Beck

The connectivity restoration in the Brancepeth Beck significantly increased the overall fish density in 2017, 2018 and 2019 relative to 2014. Brown trout, bullhead and stone loach populations benefitted from the restoration, yet the data suggest a significant ‘event’ occurred in 2014-2015 that caused the loss of all minnow and most stone loach from the upper and middle part of the stream, with recovery only starting in 2018-19. No salmon were caught – the reasons for this might be as expounded in section 5.5.1. The spawning redds found during the walkover survey provided evidence that adult sea trout managed to reach upstream of the large stepped barrier at B3 for the first time, after the construction of the rock ramp. In addition, the density of both YoY and older trout in the restored reach were markedly higher comparing with the unrestored upstream reach providing evidence that the habitat was improved for young trout in this section, and suggesting the river connectivity and passage, for trout at least, has been successfully restored in the middle reach. Unfortunately, the fish length was not measured in the 2014 pre-restoration survey, so it is not possible to know the increase in 0+ trout density across the large number of survey sites.

In 2017, all sampling sites in the restored reach were classified as Grade A for YoY trout (Table 5.11). In 2018 and 2019, six sites within the restored reach were classified as Grade A for YoY trout, two sites were classified between Grade B and D. In addition, all restored sites were classified as Grade A for older trout in 2018. Comparing with the un-restored reach, most sites were classified as Grade E or F between 2017 and 2019 for YoY trout. YoY trout was totally absent from more than half unrestored sites during the study period. Dry autumns in 2016-2018 may have caused the apparent deterioration in trout density grades over time in the unrestored upstream section (Table 5.11). Status of older trout were classified as Grade D, E and F at most un-restored sites between 2017 and 2019. Although a wide range of trout density grades (including grades D-F) was found for the unimpacted reference zone, this may be attributed to more heterogeneous flow types and low gradient stream channel.

The steadily decreased trout density in the upstream unrestored reach suggests that leaving the sites fragmented and impacted under the anthropogenic barriers would create a negative effects on the fish community and decrease the recruitment of young fish (Birnie-Gauvin *et al.*, 2020). Within the upstream reach, although B7 and B8 were “restored” by the WRT, fish were still present in a low numbers. This is mainly due to no action being taken to restore passage and reduce ponding at B5 and B6, and both barriers can still prevent fish from ascent further upstream. During the walkover survey, no spawning redd or adult sea trout were observed in upstream of B5 in both 2017 and 2018. This further proves that remaining barriers still pose significant negative effects on upstream migration. In addition, during the 2019 electro-fishing survey, it was noticed that plastic baffles on B4 were not firmly attached on the weir surface anymore and a big gap was formed between the baffle and weir. This means the easement may not be able to serve its original purpose in elevate enough water depth for salmonids to jump over the weir. It is suggested that fish pass / easement should be regularly maintained to keep its efficacy. This represents another problem with fish passes and easements; that capital spend may be provided to install them, but it is rarely provided for the careful maintenance of such structures.

Table 5.11 Brown trout classifications of Brancepeth Beck sampling sites under the EA-FCS grading scheme.

Section	Year	Species group	A	B	C	D	E	F
Reference 1-4	2017	YoY	3			1		
		Older		1	2			1
	2018	YoY	2		1		1	
		Older	2		1		1	
	2019	YoY	1	2			1	
		Older			1	1	2	
Restored 5-12	2017	YoY	8					
		Older		2	3	3		
	2018	YoY	6	1		1		
		Older	8					
	2019	YoY	6		1	1		
		Older	2	2	2	1	1	
Unrestored 13-25	2017	YoY	1	1			4	7
		Older	1		3	6	1	2
	2018	YoY		2			3	8
		Older		1	2	5	2	3
	2019	YoY			1	1	2	9
		Older			3	3	1	6

The common minnow is the only species that vanished in both restored and upstream unrestored reaches during the study period. However, this species was not caught since 2015, which is one year before the connectivity restoration. This suggests that disappearance of minnow was not related to the catchment restoration. More likely it was the result of a stochastic event such as a dry summer or poor water quality event in the upper and middle reaches. Poor water quality seems unlikely since other oxyphilic species such as trout and bullhead did not suffer in the same way. It is more likely that, since minnow are the only midwater schoolers in the stream, and rely on slack water refuges (Frost, 1943) , they were swept away during a high-flow event(s) in 2014-15 (Garner, 1997). In the downstream reference reach, minnow has been observed recolonizing the restored reach. This species managed to disperse nearly 500 m upstream in the past three years. If habitat in the restored reach keeps constant, the minnow population will be recover shortly, but may take much longer to recolonize the upper unrestored section. Stone loach also disappeared from most of the upper section at the same time, but less so from the middle section and have recovered more quickly because of that; the timing suggests they were impacted by the same event(s) but less so than minnow.

In the Brancepeth Beck, the initial connectivity restoration plan by the WRT was to mitigate these barriers from downstream to upstream. However, after they restored the first four barriers (B1-B4), they skipped two barriers (B5 and B6) in the mid-catchment and chose to mitigate barriers (B7 and B8) further upstream. In this case, both B5 and B6 are major barriers to fish passage, but both could not be removed due to different reasons. One of the problems at B5 and B6 is the existence of structures that landowners are unwilling to remove (B5) or allow to be removed (B6), even where there is no risk of flood damage to property. Another problem is cost which can be large, for example where the stream is culverted under a road. When developing barrier removal project, approaches should take into account the catchment scale to identify and prioritize the most relevant stressors affecting river connectivity (Haase *et al.*, 2013). Barrier removal through prioritization could gain a better overall connectivity increase than randomly remove barriers at the catchment scale (Branco *et al.*, 2014). In NE England, one major challenge often faced in river connectivity management is lack of funding to remove or mitigate all barriers within a catchment. Under this circumstance, mitigate barriers according to a strict model predictive basis of which should work best could be considered (Branco *et al.*, 2014), to avoid misuse of funding and ensure limited funds spend on the priorities.

In order to improve the connectivity of the upstream reach, the first and most vital step is to find an appropriate method to restore the connectivity of both B5 and B6, since both barriers are unable to be removed. The low fish density in upstream B6 suggests it is pointless to install fish passes on the further upstream barriers unless the poor passage further downstream is eased. In July 2020, a stepped stonework bypass structure was installed at B5 by the landowner. It may potentially benefit the passage of salmonids during high flows but may have less utility for poor-leaping-ability species such as stone loach, bullhead and minnow, since it is not a carefully designed fish pass. It is suggested that some modification could be done at this bypass structure, to facilitate passage for more species. Furthermore, to achieve a successful catchment scale restoration for Brancepeth Beck, the fish passes and easements need to be regularly checked and maintained, to avoid partly / completely loss of function. Although the fish abundance and community structure has benefitted from the fish pass, the river habitat was still heavily affected by the remaining weirs. If these weirs can be fully removed in the future, more suitable spawning and rearing habitat could be provided to fish, and fish such as trout and bullhead would be benefit from the restoration.

5.5.3 Cong Burn

Wide variations in trout recruitment between years and sites have been observed in the Cong Burn catchment since 2011. From a short term perspective, there is little clear signature of any positive effect from the restoration actions. It seems likely that suboptimal habitat or water quality in the lower part of the Cong Burn may be holding back the restoration of the fish community there. There is little evidence of upstream dispersal and colonisation of small benthic fish species (bullhead, loach) up the Cong Burn at this point in time. The low species diversity in S2 and further upstream sites suggesting the presence of B2 (500 m long culvert) holds back the majority of species found at S1 from colonizing upstream and caused significant negative impacts on fish communities at Cong Burn catchment. This is potentially through a fish density effect of there being sufficient dispersers to help continued upstream colonization (Radinger and Wolter, 2014). There was greater evidence of eel penetration up the catchment, but these still did not reach high densities. To achieve a successful restoration of the expected natural fish community (approximating that at S1) further upstream, the current vital step is to improve connectivity at B2. This could be achieved by replacing the culvert with a natural channel or installing nature-like fish pass elements to facilitate upstream passage. This is being progressed, slowly, but the budgeted costs are high (hundreds of thousands of pounds) and the engineering is complex, as the structures are set within an urban environment where flood risk is a primary concern and there is little space to ‘soften’ the existing hard engineering. From longer term examination in certain sites (see section 2.3.2.6), the variations in trout population indicates trout have not yet fully recolonized the river and developed dense populations that would be expected if the stream had good connectivity and good trout habitat. Furthermore, a lack of good baseline data and variability in the pre-restoration survey methods, increased the difficulty in the interpretation.

No 0+ trout (≤ 80mm) were caught at sites 9 and 10, immediately upstream of Pelton Bridge in 2016, this reflecting the output of spawning immediately upstream from autumn 2015, before the easement installation. However, it should be noted that relatively small numbers of 0+ trout were recorded in the Cong Burn in 2016 compared to the same sites in the two previous years, so the lack of trout fry in the Twizell Burn at S9 and S10 may simply reflect a poor 2015 year class in terms of recruitment. The decrease in 0+ trout density between 2014 and 2019 might be the result of worsened water quality in the Twizell Burn associated with lack of food sources, but the low autumn flows in 2016-2018, combined with the large number of obstacles, many of which have easements rather than full passage solutions, may also have contributed to the decline. In 2013, the chemical status of Twizell Burn was classified as “Fail” due to the fail in both nickel and tributyltin compounds (Environment Agency, 2020a). In 2014, the Twizell Burn failed in benzo(a)pyrene and tributyltin compounds. In addition, the status of invertebrates was classified as “Poor” in both Twizell Burn and Lower Cong Burn in 2014; then it was classified as “Moderate” in 2015 and 2016 (Environment Agency, 2020a). This is mainly caused by the organic pollution from the sewage discharge from waste water treatment.

Examining the combined data available from EA and WRT salmon have remained almost entirely absent in the Cong Burn since 2003, with the exception of a few records at S1 (see Chapter 2, Figure 2.32). This is similar to in the Deerness where salmon continue to be near-absent despite connectivity restoration actions (see Chapter 2, Figure 2.34). Salmon rarely attain high densities in small lowland streams (Environment Agency, 2002c) and, in particular, with the Wear salmon population being in a recovering state it seems unlikely that salmon will rapidly colonise such streams within the next decade. These streams may remain unfavourable for salmon (which often prefer larger tributaries and main stem habitat in which to spawn (Louhi *et al.*, 2008; Jonsson and Jonsson, 2011).

The majority stickleback population was found in the upper section of the Twizell Burn where the habitat is impacted by urbanization, where water quality is poor (waste water input from Stanley and South Moor) and where diadromous fish access is impossible due to the large barriers at Grange Villa (B9).

5.5.4 Bedburn Beck

In Bedburn Beck, both Atlantic salmon and brown trout abundance showed a decreasing trend during both short-term and long-term monitoring programmes. YoY trout density was classified as Good (Grade A and B) between 1991 and 2010 under the EA-FCS grading system, but degraded to Poor (Grade D or E) since 2011 (Table 5.12). The older trout densities were classified as Good (Grade A and B) during most years, except in 2011 and 2019 when they were classified as Fair (Grade C).

YoY salmon density varied between Grade A and D during most years, except in 2009 and 2019 when they were classified as Poor (Grade E). On the contrary, the older salmon densities were classified as Good in all years between 2001 and 2016. Until 2019, both salmon and trout abundance largely decreased, when both YoY densities were classified as Poor (Grade E), this may suggest progressively poorer recruitment. This is important, because Bedburn Beck was regarded as a ‘reference’ stream, with clean water, good physical habitat and relatively few barriers, yet juvenile salmonid densities appear to be declining. In the medium term, this reinforces the improvements in trout densities in the Deerness and Brancepeth Beck as genuine; they have improved and are recovering at a time when salmonids in one of the Wear’s ‘high-quality’ tributaries are declining. In other words this is further evidence of genuine (but incomplete) restoration success in the

Deerness and Brancepeth Beck.

Table 5.12 Brown trout and salmon classifications of the Bedburn Beck S2 under the EA-FCS grading scheme. Fish densities from 2001 to 2006, 2019 were calculated based on three-pass electro-fishing data. Fish densities from 1991 to 1997, 2007 to 2016 were calculated based on single-pass fish densities divided by mean capture efficiency from three-pass electro-fishing.

	Trout YoY		**Trout Older**		**Salmon YoY**		**Salmon Older**	
Year	Density	Grade	Density	Grade	Density	Grade	Density	Grade
1991	100.9	A	19.8	B	70.3	B	14.1	B
1995	47.3	A	16.9	B	15.8	D	22.1	A
1997	54.5	A	11.0	B	67.0	B	5.5	C
2001	50.8	A	14.8	B	29.3	C	19.7	A
2002	70.9	A	78.1	A	19.6	D	57.2	A
2003	100.2	A	138.4	A	100.3	A	214.5	A
2004	25.1	B	26.1	A	29.8	C	29.8	A
2005	65.2	A	52.9	A	37.5	C	44.4	A
2006	95.2	A	126.2	A	29.9	C	146.1	A
2007	27.3	B	19.9	B	21.6	D	72.2	A
2008	26.5	B	13.2	B	34.1	C	31.5	A
2009	30.2	B	34.0	A	4.7	E	42.4	A
2010	93.2	A	93.2	A	35.4	C	86.7	A
2011	5.7	D	6.1	C	14.4	D	20.7	A
2012	7.6	D	24.1	A	26.8	C	26.8	A
2014	30.3	B	34.4	A	43.8	C	43.8	A
2016	47.4	A	28.6	A	10.2	D	33.9	A
2019	2.3	E	7.4	C	3.2	E	4.4	D

A positive correlation between trout parr density and previous year autumn flow suggesting that trout abundance in the Bedburn Beck was highly related to the ability of spawners to access the stream, yet the lack of any such relationship for salmon is a surprise. High flow events in autumn would be expected to bring more adults into spawning streams, whereas persistent low flows in autumn could reduce the numbers of spawner in the Bedburn Beck, although low numbers of barriers would also limit such impacts.

The negative correlation between salmonids (both salmon and trout) and bullhead may suggest that competitive coexistence between these species. This is mainly due to the similar dietary niches and feeding strategy causing the competition in food resources access (Gabler and Amundsen, 1999; Floury *et al.*, 2019). The competitive coexistence of YoY Atlantic salmon parr and alpine bullhead (*Cottus poecilopus*) has been well documented in sub-arctic rivers (Gabler and Amundsen, 1999, 2010).

The fish density changes in the Bedburn Beck probably reflect long-term natural trends in fish communities, particularly of migratory sea trout populations in Wear tributaries, affected by factors such as marine survival, and river flow affecting upstream migration past barriers in the main Wear. Certainly adult salmonid counts at Durham have declined markedly in the last 6 years (Figure 2.18) and it is possible that the decline of juvenile salmonids in Bedburn Beck is an outcome of this.

5.6 Conclusions

This study assessed the effects of multiple barrier restoration works on fish communities, and findings of the study suggests that, in rivers with good aquatic habitat, including good water quality, restoring river connectivity, can be beneficial for both resident and migratory fishes. But it emphasizes the need to resolve the majority of artificial barriers in a subcatchment, rather than just a few, and that all key stressors to natural ecosystem function need to be removed in order to achieve near-complete ecological restoration outcomes. Findings of this study also support the recent emphasis of barrier removal being more effective in restoring fish communities in the immediately upstream reach compared with fish pass installation. In addition, this study showed that after connectivity restoration, the recovery periods of migratory salmonids such as sea trout could take more than three years in small moderate-gradient NE English tributaries, in contrast to the rapid recovery of migratory salmonids in low-gradient Danish streams. This study has important implications for in-stream barrier management and freshwater fisheries.

Chapter Six

General discussion

6.1 Summary of thesis outcomes

One of this thesis' aims was to determine the likely factors responsible for the decline and recovery of fish (especially anadromous salmonids) stocks in three post-industrial rivers in Northeast England (Chapter 2). Although not statistically tested, due to the nature of the evidence available, severe decline in water quality, barriers to fish movement and habitat modification all seem to have been major factors in the decline of natural fish communities in the Tyne, Wear and Tees. Removal of the most severe pollution sources has been crucial for their partial recovery, particularly for anadromous salmonids on the Tyne and Wear whose passage was inhibited by pollution conditions known to be lethal or very stressful to migrating salmonids (Bassindale *et al*., 1933; Alabaster & Lloyd, 1982). However, considerable work remains to be done in restoring water quality, habitat diversity and native fish biodiversity and abundance in all three catchments, against a background of newer impacts from intensive agriculture and invasive species.

Because anthropogenic river barriers are a key cause of river modification and impact to fish communities, including migratory species, the thesis aimed to assess the level of completeness of the current national barrier inventory using the Rivers Wear and Tees as case studies (Chapter 3). Chapter 3 revealed that 77.3% of in-stream barriers in both catchments combined were absent in the national barrier inventory, supporting the contention that the national barrier inventory is highly incomplete (Jones *et al*., 2019). Failure in restoring stream connectivity could be expected when using this incomplete national barrier inventory. The effects of removal of a single tidal barrier on aquatic habitat, fish and invertebrate communities in a lowland stream of the Tees were tested (Chapter 4) as were the outcomes of multiple connectivity restoration works on fish communities in three tributaries of moderate gradient, located in the River Wear (Chapter 5). Chapter 4 revealed that European eel (*Anguilla anguilla*) density largely increased across the catchment 1.5 years (as a result of the ability to disperse into formerly poorly accessible upstream habitat) after removal of the tidal barrier. On the contrary, brown trout density (*Salmo trutta*) remained low during the study and recruitment was poor during the short study timeframe. The outcomes of Chapter 4 support the recent emphasis of barrier removal as an effective tool to restore river connectivity and lotic fish communities.

Chapter 5 revealed that in a stream with relatively good physical habitat quality (River Deerness), both brown trout and bullhead (*Cottus perifretum*) abundance increased within the restored reach three to four years after the restoration and remained elevated until the end of the study (2019). Nevertheless, Atlantic salmon (*Salmo salar*) remain nearly absent, even though abundant in the main river. However, in a stream with degraded habitat and recovering water quality (Cong Burn), partial connectivity restoration has had limited benefits on the fish population. The results of Chapter 5 agree with the findings in Chapter 4 that wider catchment management (e.g. habitat restoration or pollution control) is required to complement connectivity restoration.

This thesis has important implications for environmental and river restoration organisations engaged in river and estuary management on the specific catchments and for elements of river restoration programmes aimed at fish stocks. It also has broader relevance to rehabilitation of fish stocks in river catchments degraded by urban and industrial development across much of Northwest Europe and parts of North America. In this chapter, a discussion of the main findings is presented along with some limitations and implications of the study, as well as recommendations for future research.

6.2 Anthropogenic impacts on fish decline in post-industrial rivers

Worldwide, many running water species are threatened with extinction and the ecosystems of many streams and rivers have become badly damaged due to anthropogenic impacts (Vörösmarty *et al.*, 2010). Although river rehabilitation schemes have been increasingly used in recent years, only a few studies assess the effectiveness of the measures on ecological characteristics from a holistic perspective (Paillex *et al.*, 2017).

Restoration ecology requires an understanding of past events and impacts in order to understand what needs to be done to restore functional elements of the ecosystem. A substantial part of this thesis was devoted to a historical review of the decline and partial recovery of the Tyne, Wear and Tees ecosystems (see Chapter 2), involving collation, integration, analysis and interpretation of fragmented evidence and data. The term

‘’Historical Ecology’’ has been used since the middle of the 20th Century for a wide range of studies of past human-nature relationships (Haidvogl *et al.*, 2015). It has been adopted as a specific method to study past ecosystems, their variations and interactions between human and environment (Hall *et al.*, 2011; Haidvogl *et al.*, 2015). Recognition of the “shifting baseline syndrome” (Pauly, 1995) in restoration ecology has resulted in an increased application of using historical data to reconstruct pre-exploitation conditions (Limburg and Waldman, 2009; Hall *et al.*, 2012; Mallen-Cooper and Zampatti, 2018). Species composition, abundance or biomass and its surrounding habitat in past ecosystems are reconstructed based on the available historical sources, and this information provides baseline reference data for restoration works (Hall *et al.*, 2011; Langford *et al.*, 2012; Mattocks *et al.*, 2017). Furthermore, study of the historical interactions between human activities and fish communities helps reveal past trends in fish community and population development, and increases the understanding of potential future trends (Langford *et al*., 2012; Haidvogl *et al*., 2014). However, historical studies of river and freshwater fish ecology, as attempted within this thesis, are still rare.

One reason for this is due to the difficulties in gathering historical biodiversity data (Haidvogl *et al.*, 2015). Studies of historical fish communities or populations often have incomplete data such as abundance, biomass and even species presence (Limburg and Waldman, 2009; Haidvogl *et al.*, 2015). Beyond the last century (and even more recently in many cases, as in Chapter 2), freshwater fish surveys often relied on interviews with fishermen (Haidvogl *et al.*, 2015). Statistical data on wild fish catches were rarely recorded in the medieval period (Hoffmann, 2015) and even much later as in this thesis (Chapter 2), and most fish community and population data were produced in relation to commercial or traditional fisheries, fish trading and fish consumption before the 1950s (Limburg and Waldman, 2009; Hall *et al*., 2012; Haidvogl *et al*., 2014, 2015). However, these data were mostly focused only on species of commercial interest and data were often not well preserved or even lost (Haidvogl *et al.*, 2015). It was not until the second half of the 20th Century, when standardized and systematic fish sampling approaches were largely employed in monitoring programmes (Haidvogl *et al.*, 2015). Again, these patterns are reflected in Chapter 2, as the only readily accessible fish abundance data or proxies of abundance (fish catch) in the Tyne, Wear and Tees pre-1990, were primarily for Atlantic salmon and sea trout (*S. trutta*), representing well under 10% of native fish species in these rivers.

In recent years, rather few case studies on historic changes in fish species or communities of European rivers have been published (Worthington, 2010; Haidvogl *et al.*, 2015; Hoffmann, 2015; Torkar and Zwitter, 2015; Wolter, 2015). In the second half of the 19th Century, the Industrial Revolution led to large-scale and frequent exchange of species (Haidvogl *et al.*, 2015). For Northeast English rivers, such changes mainly resulted in temporary or permanent loss of species such as Atlantic salmon and smelt (*Osmerus eperlanus*). Although some fish species were recorded in historical literature, the detailed records are too poor to be clear as to what the original natural fish faunas were. The Northeast English rivers exist in a biogeographical transition zone between the Humber system to the south (tributaries of the Swale in the Humber system lie just a few hundred metres from those of the Tees) and the salmonid dominated rivers of eastern Scotland (Davies *et al.*, 2004). At the end of the last Ice Age the Humber was linked to the Rhine system and shared a range of cyprinid, gadid (burbot *Lota lota*), percid and esocid species generally not found in rivers further north in Britain. It is likely that many of the cyprinids, as well as pike (*Esox lucius*) and perch (*Perca fluviatilis*) found in the Tees were introduced from the Humber system many centuries ago. Barbel (*Barbus barbus*) in the Wear are a recent introduction in the last few decades (Britton and Pegg, 2011; McParlin, 2011). Grayling (*Thymallus thymallus*), roach (*Rutilus rutilus*) and dace (*Leuciscus leuciscus*) are all known successful introductions to the Tweed whose headwaters border those of the North Tyne (Mills, 1989; Dawnay *et al.*, 2011). It is likely, but less clear cut, that these and several other fish species are non-native to the Tyne and Wear.

In river systems, fish are one of the major biota that were affected strongly during the Industrial Revolution, partly because of their sensitivity to altered ecological conditions, but partly also because at least some of them were readily exploited by humans. Understanding the key factors that led to the decline of fishes in industrial rivers and their subsequent recovery, or not (e.g. Atlantic salmon, Wear vs. Tees) can help guide management plans for future conservation, and provide some insights before conducting river rehabilitation works. For example, historical literature was used to characterize the historical decline, distribution, and environmental requirements of burbot in England as an indicator of the utility for planning conservation actions such as burbot reintroduction (Worthington, 2010; Worthington *et al.*, 2010). In order to do that, the former distribution of burbot populations have been reconstructed in 42 rivers in eastern England (Worthington, 2010). However, there had been no systematic historical study on the decline and recovery of fishes in post-industrial rivers in NE England, though this thesis has attempted to remedy that.

Due to the limitation in gathering historical fish data before the 19th Century, Chapter 2 briefly described the status of fishes in the Tyne, Wear and Tees before the 19th Century and mainly focused on the changes from the 19th Century to the present day. Chapter 2 described the deterioration of water quality caused by intensive mining in the North Pennines, industrial pollution and domestic waste since the 19th Century, then it described the connectivity and habitat deterioration caused by damming actions (e.g. many old mill weirs as well as more recent barriers such as Kielder reservoir and the Tees Barrage) and intensive gravel extraction. Later it showed the massive decline in anadromous fish populations (primarily salmonids) in the three catchments. Before the 19th Century, Atlantic salmon and sea trout were widely distributed through the Tyne, Wear and Tees catchments. However, multiple pressures from human activities led to the functional extinction of salmon and sea trout from all three rivers in the early 20th Century. The near absence of salmon and sea trout lasted until the 1970s, then their numbers steadily increased in all three catchments as water quality improved. In the Tyne and Wear catchments, the increase of salmon and sea trout has slowed down and stabilized in recent years. There is even some evidence of a recent decline in the Wear. In the Tees, salmon and sea trout numbers have slowly increased until today, rather than following the rapid trajectory of the Tyne and Wear. This is potentially caused by the Tees Barrage, constructed in the estuary in 1994, and which prohibits free upstream and downstream migration of fishes.

Chapter 2 has revealed that the potential for recovery of anadromous salmonid stocks in post-industrial Pennine rivers with abundant salmonid spawning and nursery habitat, is driven by both accessibility and survival in the lower river, through the effects of barriers, pollution and predators (e.g. human). The Tyne and Wear are now England's most productive salmon rivers. By contrast, salmon populations in the River Frome and other chalk rivers in the south of England have continuously declined since the 1970s (Welton *et al.*, 1999a; Ikediashi, 2015; Cefas *et al.*, 2019). Furthermore, although lots of effort was made to restore river habitat and connectivity, salmon failed to recolonize in the River Thames in the south of England (Chapter 2). Salmon populations in the south of England are at risk, or unable to recover, due to climate change, water abstraction, intensive farming impacts and urbanization (O'Neill and Hughes, 2014; Cefas *et al.*, 2019). By contrast those in the north of England are less susceptible, currently, to climate change and hydrological pressures.

Chapter 2 was mostly focused on the anthropogenic impacts on the river systems and salmonid populations since the 19th Century. However, the knowledge of past fish communities, populations and habitat conditions in Northeast England rivers prior to the 19th Century are still poorly understood. Although some potential useful archived documents were noticed (including several documented within Skelton, 2017), there was no access to these archives due to the lockdown during the global coronavirus pandemic. Future research on this topic might better exploit analysis of archaeological finds, fishery board reports, estate management records (e.g. accounts, rentals, court books, etc.) and commercial / recreational fishing records (e.g. fish trading or angling club logbooks) (Hoffmann, 2015). For example, in-stream habitat and aquatic faunal community could be reconstruct by using the sub-fossil remains of aquatic invertebrates (Greenwood *et al.*, 2003; Seddon *et al.*, 2012). However, these types of research may require interdisciplinary cooperation between ecologists, archaeologists, hydrologists and historians (Haidvogl *et al.*, 2015).

Although the most severe pollution sources in the Tyne, Wear and Tees have largely been reduced (especially those linked with heavy industry and sewage), and a clearly increasing trend of salmonid and estuarine fish abundance has been observed in the three catchments, these rivers are still facing great challenges in some aspects of their management. Only a small proportion of the Tyne (33.3%), Wear (9.4%) and Tees (14.9%) Water Framework Directive (WFD) waterbodies had achieved good ecological status by 2019. All three catchments are still affected by multiple anthropogenic impacts such as physical modifications (e.g. instream barriers, channelization), point source pollution from abundant mines, sewage treatment works, and diffuse pollution from rural areas and towns, cities and transport. All these issues need to be assessed by river management agencies in the near future and the prioritization of what types of restoration work should be done, and where, need to be carefully evaluated.

The results of Chapter 2 provided baseline water quality and fish community background information of multiple study sites including all those in Chapters 4 and 5. Providing accessible and transparent data and information is crucial to long-term ecological study such as river restoration (Lindenmayer *et al.*, 2012; Powers and Hampton, 2019) and is something it is hoped this thesis will help with in regards to future restoration actions on the study catchments. One of the frustrations of this study has been that although it was known that fish community / abundance and estuary water quality sampling were carried out within the three catchments in an orderly sampling programme from the 1970s onwards, accessing those data was extremely difficult. In many cases it seems that the organizations responsible (Northumbrian Water Authority) and their successors were poor at archiving and making data available. Even for the National Rivers Authority and its successor Environment Agency, some of their early stage electro-fishing data were only preserved with hard copies and have not been digitised. So, getting data from the early 1990s was reliant on individual contacts, rather than accessible long-term databases of records. This problem is not new (Langford *et al*., 2012), but environmental managers need to better value, archive and make accessible long-term monitoring data sets, rather than to allow them to be ‘lost’ when regulatory or monitoring bodies are reorganized.

Another issue of the Environment Agency and third sector bodies such as the Rivers Trusts is their long-term electro-fishing monitoring plans are often badly organized or appear ad hoc. When analysing the electro-fishing data, it was noticed that within a catchment, sites were often sampled in a disordered way with bad temporal continuity (Defra, 2010b). The same sites were not surveyed in each year (yet fish recruitment can vary widely between years), some sites were only surveyed once or twice and then were abandoned for the remaining period. The electro-fishing method varied between years and survey sites: some were surveyed quantitatively with several passes and stop nets used, others were single passes with or without stop nets (Defra, 2010b). This is potentially due to limited funding or staff. However, badly organized sampling spatially and temporally, increased the difficulty in data analysis and limited the data utility in statistical analysis. It is suggested that when developing future electro-fishing surveys, the locations and numbers of sampling sites within the catchment need to be carefully planned to avoid mis-spending large sums of money, and sites should be surveyed repeatedly (e.g. on a yearly basis) to facilitate detailed statistical analysis. Consistent, annual quantitative sampling of fewer sites is probably of greater value in tracking the condition of rivers being restored than multi-year interval sampling of a greater number of sites. In future, environmental DNA sampling may also prove increasingly valuable (see section 6.4 for a discussion of its utility), but again any transition of sampling method needs to be done carefully and rigorously.

6.3 Instream barrier management

Chapter 2 revealed that the presence of instream barriers is still one of the major reasons for not achieving good ecological status in the three study catchments (Tyne, Wear, Tees). Instream barriers can alter habitats, ecosystem processes and restrict or prevent fish migration and dispersal, and eventually lead to a decline in the abundance of sensitive species and a decline in biological diversity (Chapters 4 and 5). Results of Chapter 4 revealed that although a small tidal barrier did not fully prevent upstream eel passage, it dramatically reduced eel abundance and altered eel size structure within the upstream reach. In addition, fish density in the impounded reach was significantly lower compared with other sections prior to barrier removal. Numerous similar observations have been made elsewhere, although the exact patterns of effects varies with barrier location within a catchment, proximity and density of other barriers, river gradient and topography and native fish community (Gehrke *et al.*, 2002; Katano *et al.*, 2006; Ding *et al.*, 2018; Galib *et al.*, 2018).

In order to achieve good ecological status for as many constituent lotic waterbodies as is feasible in all three study catchments by 2027 (the WFD deadline which the UK, although having left the EU will follow), one of the major catchment management tasks is to reduce the ecological impacts of all artificial instream barriers, and restore river connectivity. Anthropogenic barrier management, such as barrier removal and fish pass construction, has been increasingly used in river connectivity restoration programs. However, it is difficult to quantify the degree of habitat fragmentation in a catchment due to incomplete barrier records, duplicate databases, out of date information and other issues. To understand the degree of river fragmentation, it is necessary to have a complete barrier inventory including barrier distribution, type, height, slope and other parameters across all stream orders. Creating a complete river barrier inventory is a priority step in developing the barrier management processes such as barrier removal or modification (Atkinson *et al*., 2018; Belletti *et al*., 2020). In order to do that with minimal error, the whole river channel needs to be assessed in the field, location of all potential barriers need to be recorded, features of each barrier need to be measured, then the information need to be categorised and integrated into a comprehensive barrier inventory.

Although having a complete instream barrier inventory is necessary before conducting catchment-scale restoration works, many countries do not have such barrier inventories and for most countries that have a barrier inventory it is very incomplete (Belletti *et al*., 2020). For example, in Denmark, low-head weirs and culverts were missing from the Danish national barrier inventory (Birnie-Gauvin *et al.*, 2017a), and the United States National Inventory of Dams only records dams higher than 10 m and ignored all low-head dams (US Army Corps of Engineers, 2020). In England and Wales, a national river barrier inventory was produced as a byproduct of a hydropower potential survey (Entec, 2010), and is held, updated and managed by the Environment Agency of England and Natural Resources Wales respectively. The study described in Chapter 3 (Wear and Tees catchments) revealed that 77.3% of in-stream barriers in both catchments combined were absent in this national barrier inventory, including 68.6% of artificial barriers and 82.6% of natural barriers. It is estimated that there are nearly 500 instream artificial barriers still present in the Wear catchment and 600 artificial barriers in the Tees catchment. All varieties of artificial barriers were found to be absent from the database. If environmental agencies (in this case, the EA) use national barrier inventories for connectivity restoration without field verification, it may lead to inefficiencies in restoration, or waste of effort.

One example is that during the barrier survey (Chapter 3), several unidentified barriers were recorded for the first time in the Priest Burn, a tributary of the River Deerness, one of the major study streams in Chapter 5. This revealed that the River Deerness sub-catchment had not been fully surveyed by the Wear Rivers Trust or Environment Agency, before conducting connectivity restoration works, despite it being the subject of a half million pound connectivity restoration project (funded by the DEFRA "Catchment Restoration Fund") between 2012 and 2015. The presence of these barriers may lead to inefficient restoration in the Priest Burn, and may also explain the reason why trout density at sites in the upper parts of Priest Burn ("Cornsay Colliery restoration sites") did not increase during the study period, even though initial upstream recolonization by several fish species occurred soon after a road culvert easement at Cornsay Colliery (Tummers *et al.* 2016).

In some cases, using low-resolution maps during the data collection stage may lead to an incomplete database, because barriers located in small headwater streams, or side channels may be ignored. For example, the European Environment Agency catchments and rivers network system (ECRINS) dam database records barriers using the 1:250,000 resolution map. Strahler first-order streams on a high-resolution map (e.g. 1:25,000) would not appear on the ECRINS map and if an artificial barrier is located in such low-order headwater stream, it may not be able to be mapped in the ECRINS database. A previous barrier assessment study conducted by Jones *et al*. (2019) surveyed 19 rivers across the UK, but used the coarser resolution of the ECRINS river network which means the first-order streams recorded in their study were actually of a size considered as second-order streams in my study (Chapter 3). Despite this, the extent of under-representation of small anthropogenic barriers across river catchments by Jones *et al.* (2019) and in Chapter 3 were broadly comparable. In Australia, a desktop GIS analysis study identified 5,536 potential barriers in the wet tropics region (Kroon and Phillips, 2016). However, first-order streams were excluded from their study and analysis was conducted on a 1:100,000 scale map, which potentially makes the barrier inventory less useful in river management (Atkinson *et al.*, 2018). Overall, 78% and 57% of artificial instream barriers were located in first- and second-order streams combined in the Wear and Tees catchments (Chapter 3). Because many anthropogenic barriers exist in small-sized channels, whether urban or rural, it is important to map these at as large a scale as is possible; given that easily accessible GIS databases exist at 1:25,000, this would be the minimum resolution that seems appropriate, at least for well mapped regions such as Europe.

Results of Chapter 3 have revealed that first-order streams comprised about 40-50% of total river network length in the Wear and Tees catchment at 1:25,000 scale. Headwater streams (including both first- and second-order channels) are a fundamental part of any river catchment (Richardson, 2020). In recent years, headwater streams have received increasing attention due to the critical roles they play in contributing to productivity and the integrity of downstream ecosystems (Richardson, 2020). Headwater streams contribute to the characteristics of the downstream network with regard to water quality, sediment transport, and organic matter supply (Wipfli *et al.*, 2007; Church, 2015; Richardson, 2020). Besides, these streams can provide breeding and rearing habitat for fish species or ecotypes (e.g. sea trout) that eventually form part of downstream biological communities (Aarestrup *et al.*, 2003; Richardson, 2020). So, these head water streams should not be excluded from barrier surveys, and the spatial resolution for barrier audits needs to take careful consideration of the environmental restoration objectives.

The outcomes of Chapter 3 should help directly in managing future connectivity restoration in the Wear and Tees. The barrier data collected during the walkover surveys were integrated with several other databases including the national barrier inventory, the Environment Agency North East obstructions database, Obstructions All NEA database and Wear and Tees Rivers Trusts databases to provide single up-to-date inventories for each river. This has generated the first intensive but, as yet still incomplete, inventory of artificial and natural barriers in the Wear and Tees catchments. This barrier inventory and associated barrier photographs (links available in Appendix I) were shared with the Wear Rivers Trust, Tees Rivers Trust and Environment Agency, providing a valuable resource for river restoration work in the future. Of course it is crucial that this is kept up to date, that new barriers are added as they are discovered, that barrier removal and fish passage information is added also. There is no current agreement as to who should maintain catchment barrier databases in the UK; where rivers trusts exist, they are well placed to do this, but do not currently receive funding for this purpose.

In recent years, some projects were developed across the EU for developing standardized methods to record and manage river barriers. For example, within the UK, the River Obstacles project was developed by the Scottish Environment Protection Agency (SEPA), the Rivers and Fisheries Trust for Scotland (RAFTS), the EA and the Nature Locator team. A mobile ‘app’ was developed for citizen scientists to enter information on barriers found, and guidance provided to enable the collection and use of information on the location and type of artificial and natural barriers in rivers across the UK. Similarly, the AMBER (Adaptive Management of Barriers in European Rivers) project collaboration with 20 partners from 11 European countries, compiled a river barrier database, ‘The Pan-European Atlas of In-Stream Barriers’ for hundreds of rivers within each of the European countries, in order to apply adaptive management to the operation of barriers in European rivers so as to achieve more effective and efficient restoration of stream connectivity (Belletti *et al.*, 2020).

Due to the intensive, time-consuming field survey process and difficulties in getting land access permission, it was impossible to survey every single tributary of the Wear and Tees catchments during this PhD study. The barrier survey project was limited by surveying 32.8% of total stream length of the two study catchments combined, but done in a way that stratified surveying across stream orders, altitudes, land use types and locations within the catchments. In order to develop a comprehensive national barrier inventory, it is recommended that future barrier surveys are undertaken in all subcatchments and covering all stream orders. However, due to logistical reasons and the amount of effort needed, it is not possible to conduct walkover surveys for each single river by only river management practitioners. Other survey methods including recruiting trained volunteers, using low-cost aircraft such as drones (Ortega-Terol *et al.*, 2014) and encouraging citizen scientist records with phone apps (e.g. AMBER Barrier Tracker app and River Obstacles app) should be considered (Atkinson *et al.*, 2018; Jones *et al.*, 2019).

Another big challenge in barrier management in the UK is due to the nature of land ownership. Theoretically, when developing barrier removal projects, approaches should take into account the catchment scale to identify and prioritize the most relevant artificial barriers affecting river connectivity (Haase *et al.*, 2013), but this is often not the case in real life. Scientists may aim to apply theoretical principles to river restoration, but river managers are forced to work in practicalities (Dufour and Piégay, 2009). In England, many instream barriers or river sections are privately owned rather than state-owned, and in many cases, the ownership of barriers is unknown or contested. So barrier mitigations or removals frequently occur at sites where there is greatest facilitation by stakeholders and owners, not necessarily at the highest priority sites in restoration terms. For example, the Wear Rivers Trust chose to improve stream connectivity at a pipe culvert ford located in the middle reach of Brancepeth Beck. However, two other major barriers, located ~1km further downstream cannot yet be removed due to objection from the landowner and because of technical difficulties; these barriers still pose significant problems for upstream fish movement (Chapter 5).

6.4 Effects of connectivity restoration

Under the river restoration context, it is often accepted that return of a river to some condition present prior to human alteration is commonly impossible (Dufour and Piégay, 2009; Wohl *et al.*, 2015). Firstly, the past environmental context that resulted in former river processes and forms no longer exists (Wohl *et al.*, 2015). Secondly, knowledge of past river conditions is insufficient to support restoration. Finally, river systems follow complex trajectories that commonly make it impossible to return to a previous state

(Dufour and Piégay, 2009; Wohl *et al.*, 2015).

All over the world, legal mandates often require that a degraded aquatic system be restored to some degree, measured by either the status of aquatic and riparian habitat and/or the condition of biota (e.g. the EU Water Framework Directive) (Beechie *et al.*, 2009). Under this river management context, one question often raised by the restoration practitioners is: “How do we know we have restored enough?” (Beechie *et al.*, 2009). In order to answer that question, a comprehensive monitoring scheme would be needed, to assess if the restoration work has met the intended objectives, and to draw generalized conclusions about the project effectiveness (Palmer *et al.*, 2005; Beechie *et al.*, 2009).

To restore lotic habitat and river connectivity, river barrier removal has been increasingly used as a conservation tool, but evidence of its efficacy is incomplete. Two studies described in Chapters 4 and 5, used fish communities as an indicator to assess the effectiveness of connectivity restoration. Results of these two studies revealed that barrier removal was effective in restoring longitudinal connectivity, and beneficial for both resident and migratory fishes when the habitat condition is good. In Chapter 4, the rapid recovery of the eel numbers (by way of relatively unimpeded dispersal from the Tees estuary) was observed at all upstream sampling sites in the second autumn after the weir removal. However, for European eel, which spawn in the Sargasso Sea, but use freshwaters (and coastal waters) across Europe to grow, any recovery in population abundance can only truly be measured when integrated across the whole distributional range. Nevertheless, this does not diminish the evidence of benefit to eel dispersal to habitat made available by removal of the barrier described in Chapter 4. On the contrary, the brown trout population response of Claxton Beck can be measured at that spatial scale due to their philopatric tendency, but trout did not increase during the short study period. Because of lag times in natural recolonization, the prolonged generation time of fishes (several years) and due to natural stochasticity in reproductive success in fishes, high-quality studies of the effectiveness of restoration need to last decades, be on a sufficient spatial scale (likely the extended reach, subcatchment or catchment scale) and incorporate a control subcatchment. Published studies of this type are currently non-existant, but are something we need to aim for.

Because monitoring a single metric can be insufficient to determine whether restoration action has achieved its objectives (Beechie *et al.*, 2009), in order to conduct a holistic assessment, the Claxton Beck tidal barrier removal study (Chapter 4 monitored the changes of aquatic habitat and invertebrate communities in addition to fish community changes in response to the connectivity restoration. Results of the study showed that river habitat diversity increased immediately upstream and remained similar downstream following barrier removal. In addition, changes to macroinvertebrate communities occurred upstream and downstream of the former barrier but these were minor and transient. Therefore this can be regarded as having been a successful restoration, even though additional management actions are needed further upstream in the intensively farmed upper and middle reaches, to improve conditions for diverse stream and riparian habitat and to support recolonization by sensitive indicator biota such as brown trout.

Chapter 4 studied the effects of single barrier removal on the fish community, habitat and invertebrates. However, in most cases, one river system or tributary is often fragmented by multiple barriers (Chapter 3), and very few studies have been published on evaluating the effects of multiple barrier removal on remediating aquatic communities over the whole catchment / subcatchment. In some highly fragmented river system such as the Cong Burn in the River Wear catchment, more than 30 instream barriers need to be removed, in order to achieve complete connectivity restoration. In Chapter 5, multiple connectivity restoration projects were conducted in three tributaries in the Wear catchment with an attempt to restore the aquatic connectivity and fish populations. Different outcomes were observed between these three streams. In the River Deerness, and within the connectivity-restored reach of Brancepeth Beck, both brown trout and bullhead benefitted from the restoration and their abundance increased. Trout abundance at the majority of connectivity-restored sites achieved excellent or good status under the EA-Fisheries Classification Scheme grading system. In contrast, in the connectivity-unrestored reach of Brancepeth Beck, trout density even gradually decreased during the study period. Results of Chapter 5 have also revealed that migratory fish may not immediately respond to restoration works, delays may be expected when the existing fish population in the river is small and/or the generation time is long. So understanding the time lags in recovery processes and incorporating knowledge of these time lags into river monitoring programs is important. However, in these tributaries no adequate fish abundance data prior to the construction of these in-stream barriers existed, so we will never know to what extent does the fish recovered from the restoration. Use of reference conditions (in this case non-degraded streams with good water quality, diverse habitat and no artificial barriers) are the benchmark against which full restoration might be assessed (Palmer *et al.*, 2005) but the reality is that in the Northeast English rivers under study almost no such streams exist. Findings for Chapters 4 and 5 suggest that when developing future fish community surveys, standardised long-term monitoring is needed in order to assess the recovery of fish community under longer periods.

One of the frustrations in the Chapter 4 study was that land access to the very upper section of the Claxton Beck catchment was prohibited due to road construction and residential development works through the study period. The electro-fishing surveys could only be conducted in the remainder of the catchment. Another frustration is that during the pre-restoration monitoring program in the Brancepeth Beck, none of the fish lengths were measured (Tummers, 2016). So it was not possible to know the changes in YoY trout abundance (a direct indicator of year-to-year spawning success) in response to the connectivity restoration. Similar issues were observed in the EA electro-fishing dataset, where no coarse fish lengths were measured during electro-fishing surveys, which makes their data have less utility. It is suggested that, when developing a fish monitoring programme, at least 50 fish per species should be measured (and/or aged) randomly at each sampling site to generate demographic information.

Electro-fishing is, in much of the world, the conventional quantitative method for sampling fish in streams (Reynolds and Kolz, 2013). Despite its effectiveness for the collection and enumeration of many fish species it is labour intensive, ineffective in high flows or turbid water and samples relatively short stream sections (Reynolds and Kolz, 2013). In five years of post-restoration sampling at over 20 sites in the River Deerness no salmon were recorded in samples. Does this mean salmon were absent from the connectivity-restored section of the Deerness? Possibly, but in handling thousands of trout, including 50 mm long fry, it is quite possible that some salmon were missed, or even misidentified as trout due to trout being vastly more abundant. Environment Agency electro-fishing data recorded very low densities of salmon, once only, in 2011, in the Deerness close to the confluence (Figure 2.34).

Could other methods be more effective than electro-fishing in measuring the fish community locally, quantitatively and more efficiently? In recent years, the method of using environmental DNA to monitor certain fish or other aquatic animal species in streams has been increasingly developed (Atkinson *et al.*, 2019; Bracken *et al.*, 2019; Antognazza *et al.*, 2020). This method detects species through analysis of eDNA (shed from cells, from mucus etc) in water samples, requires less sampling effort but considerable laboratory skill and processing, and is currently more expensive compared with three-pass electrofishing (Bracken *et al.*, 2019), but costs are decreasing and analysis throughput is increasing rapidly. In some cases, this approach has also been applied to assess the response of fish species following instream barrier removal (Duda *et al.*, 2021; Muha *et al.*, 2021). Compared with traditional fish sampling methods, the eDNA approach is more efficient in targeting species that exist at low densities or difficult to catch using conventional methods (e.g. eel, lamprey and crayfish), or in turbid water conditions (Atkinson *et al.*, 2019; Bracken *et al.*, 2019). For example, in Brancepeth Beck no minnows were found at the most upstream sites during the 2017 survey, and with the traditional electro-fishing method < 10% of stream length could be sampled, but using the eDNA sampling approach could give greater sensitivity to detect this species even where rare, potentially integrating DNA from a significant length upstream. However, the eDNA method still has some limitations, because it can only give measures of relative fish abundance rather than actual fish density, and the method cannot provide information regarding age or size structure (Bracken *et al.*, 2019). Nevertheless, it is suggested that under certain circumstances traditional electro-fishing methods could be combined with eDNA sampling approach, to obtain a more robust result. In the future river restoration evaluation will undoubtedly make greater use of the method.

6.5 Concluding remarks

This PhD study has contributed information relevant to freshwater restoration ecology. Firstly, it identified likely drivers that led to the decline and recovery of fishes in post-industrial rivers in Northeast England. This study attempted to combine all sources of information with regard to the impacts of anthropogenic activities on aquatic habitat and fishes in post-industrial rivers. Findings of the study could support management plans for future river conservation and restoration works. Secondly, this study identified key defects in the current national river barrier inventory, and suggests that partial or complete failure in restoring stream connectivity may be expected if an incomplete barrier inventory is used. Finally, this study assessed the effects of small in-stream barrier mitigation and/or removal on fish communities in streams. These findings suggests that barrier removal, when carried to ‘best practice’ guidelines should be considered as an effective method in restoring river connectivity. Results of this study may be of help in mitigating river fragmentation, developing effective river barrier management plans and protecting endangered fish species.

Appendices

Appendix I: Supporting information for Chapter Three

This dataset is an inventory of artificial and natural instream barriers in the Wear and Tees catchments. It provides detailed information, including barrier location, types, height etc… Any in-stream structure having a vertical or steeply sloping (>45°) step, exceeding 0.2 m in height (0.5 m for natural obstacles), was recorded as a potential barrier. Additional shallow-depth barriers (e.g. culverts) were also recorded. Potential barriers are recorded as natural or artificial. Note that in watercourses in the upper catchment that have been subject to mining, some barriers recorded as 'natural' (waterfalls, cascades), although appearing of relatively natural form, are likely to have been generated as a result of human activities such as 'hushing'. We also recorded sites where barriers had existed in the recent past (EA database) but had collapsed, breached or been removed deliberately within the areas surveyed. Sections of river subject to walkover (RS/SG) and GoogleEarth view are detailed in the paper below. Photographs are available by fileshare for Durham walkover sites according to the ID codes given in database; photographs are also available separately for a proportion of EA database barriers within their in-house regional database (contact them directly). Dataset and photographs are available at: https://www.dropbox.com/sh/h9j8u98jctarn8d/AADwZur5YkKxwHeJguIO3hIta?dl=0

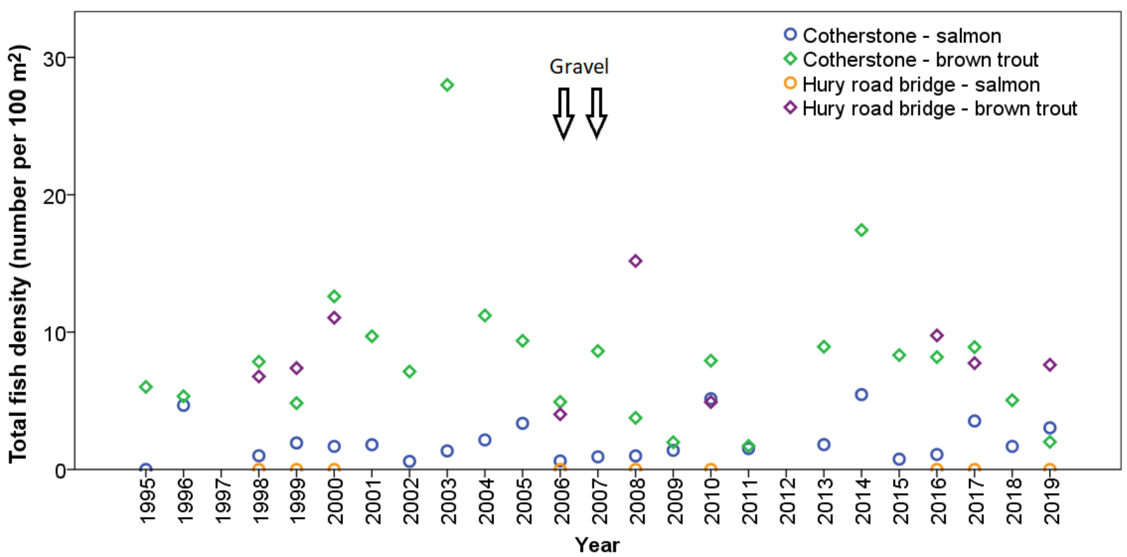


Figure S3.1 Juvenile salmonid densities (n/100m^2) below the Hury Reservoir in the River Balder, before and after the gravel reintroduction.

Appendix II: Supporting information for Chapter Four

Examples of R-scripts:

```
## LMM analysing the changes in fish density before and after the barrier removal.
Env<-read.csv('Fish.csv')
attach(Env)
library(lme4)
library(lmerTest)
Env$section<- factor(Env$section)
Env$site<- factor(Env$site)
Env$season<- factor(Env$season)
str(Env)
LLM<-lmer(fish~period+(1|season)+(1|section/site), data=Env)
anova(LLM)
```

```
##PERMANOVA analysing the changes in fish community between Period 1 and 3.
Fish<-read.csv('FSP1P3.csv')
Env<-read.csv('ESP1P3.csv')
attach(Fish)
attach(Env)
library(vegan)
range(Fish^.25)
betad <- betadiver(Fish, "z")
Env$Year<- factor(Env$Year)
Env$Section<- factor(Env$Section)
Env$Site<- factor(Env$Site)
str(Env)
adonis2(betad~Period, data=Env, strata="Site", perm=999)
```

```
##NMDS - fish communities at Period 1 and 3.

Fish<-read.csv('FS123p1p3.csv')

Env<-read.csv('ES123p1p3.csv')

attach(Env)

Year<- factor(Env$Year)

Section<- factor(Env$Section)

site<- factor(Env$site)

range(Fish^.25)

Sp_dist <- vegdist(Fish^.25, method="bray", trymax=999)

nmds <- metaMDS(wisconsin(sqrt(Sp_dist)))

nmds

plot(nmds, type='t')

op<-ordiplot(nmds, type = 'n', cex.axis=1.25, cex.lab=1.5)

pch_site = c(1, 5, 6)

col_loc = c('red', 'cornflowerblue', 'chartreuse4')

col_ellipse = c('pink', 'pink','cyan', 'cyan','green','green')

ordiellipse(nmds, draw = "polygon", groups <- factor(SY), col = col_ellipse, lty = 'dotted')

ordispider(nmds, groups = SY, col=c ('blue','red', 'blue','red','blue','red'), lwd=2)

points(nmds, lwd = 2, cex = 1.25, pch = pch_site [Env$Section], col =
col_loc[Env$Section])
```

Figure S4.1 Photo montages of habitat surveyed sites downstream of the weir before and after the removal.

Figure S4.2 Continued photo montages of habitat surveyed sites downstream of the weir before and after the removal.

Figure S4.3 Photo montages of habitat surveyed sites upstream of the weir before and after the removal.

Figure S4.4 Continued photo montages of habitat surveyed sites upstream of the weir before and after the removal.

Appendix III: Supporting information for Chapter Five

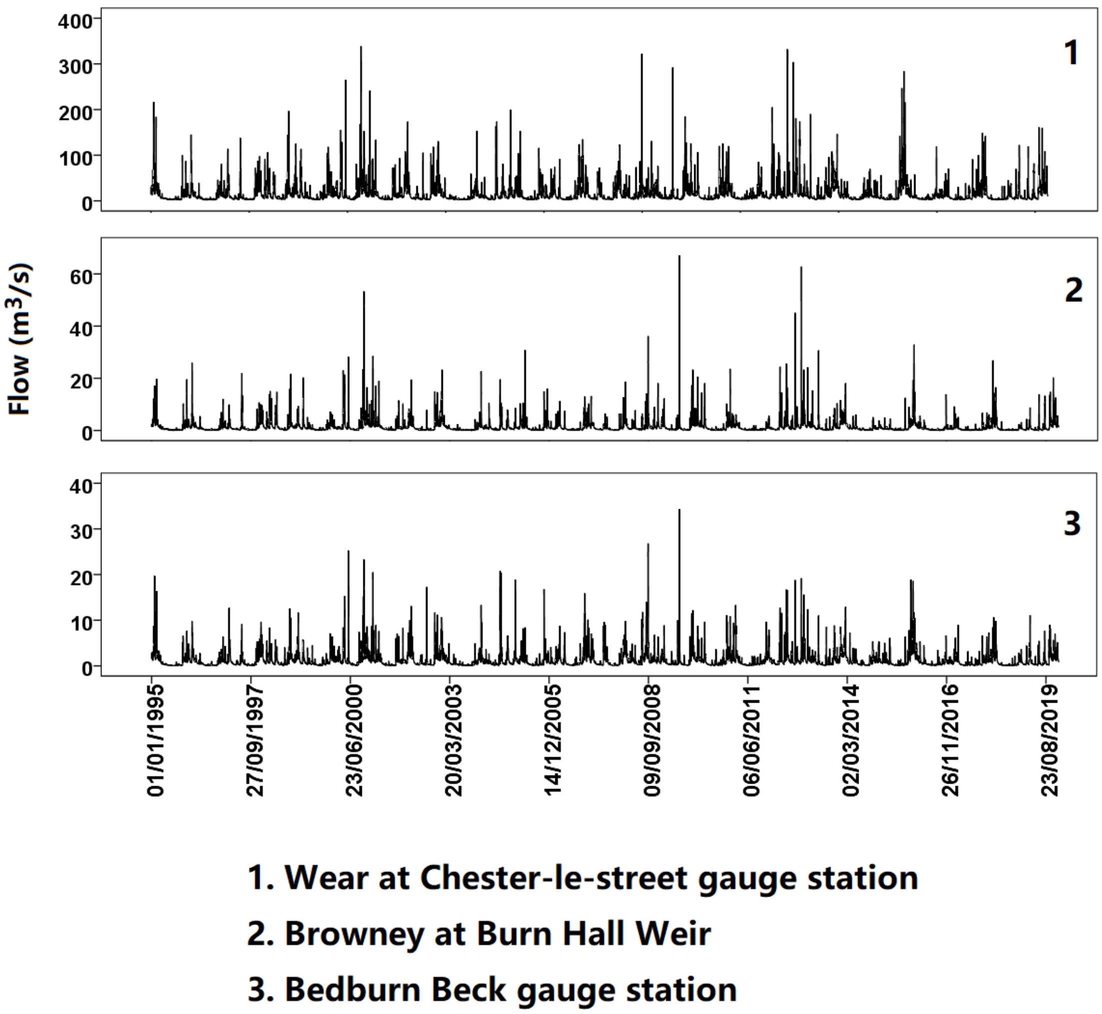
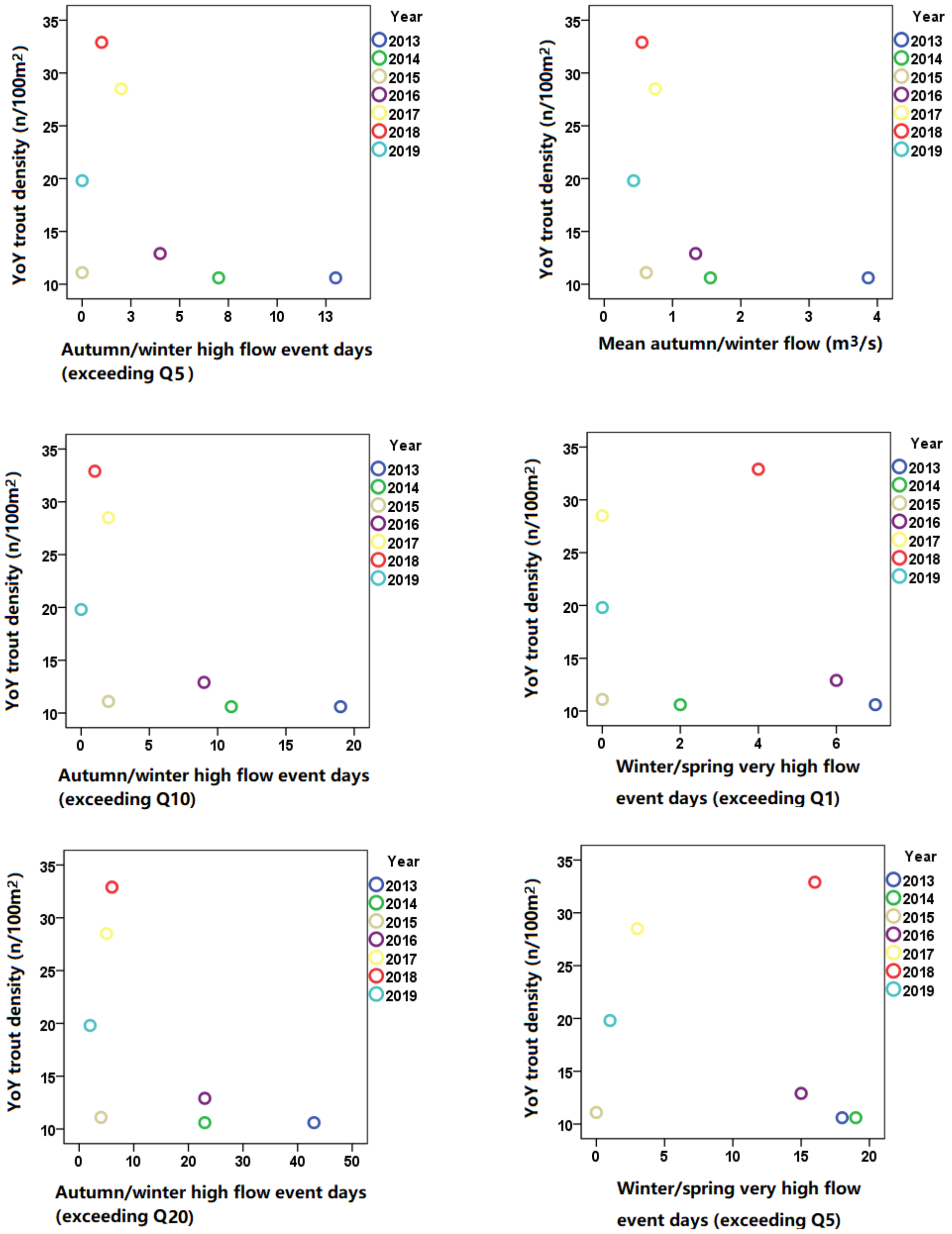


Figure s5.1 Mean daily flow (m/s^3) at Chester-le-street guage station in River Wear; Burn Hall flow guage station in River Browney and Bedburn Beck guage station in Bedburn Beck between 1995 and 2019.



Autumn/winter periods : 1 September - 15 December
Winter/spring periods: 16 December - 15 May

Figure s5.2 Relationship of YoY trout density and mean daily flow / high flow event days in the River Deerness between 2013 and 2019.

After the baffles re-construction and rock ramp installation at B7, flow velocity at the entrance of the bridge apron significantly reduced and water depth under the bridge culvert increased more than 10 cm (Figure s5.3). At Newfield Bridge site, the gabion mattress was now buried under the boulders. A slow flowing deep glide has been shifted to a shallow and fast flowing riffle habitat (Figure s5.4).

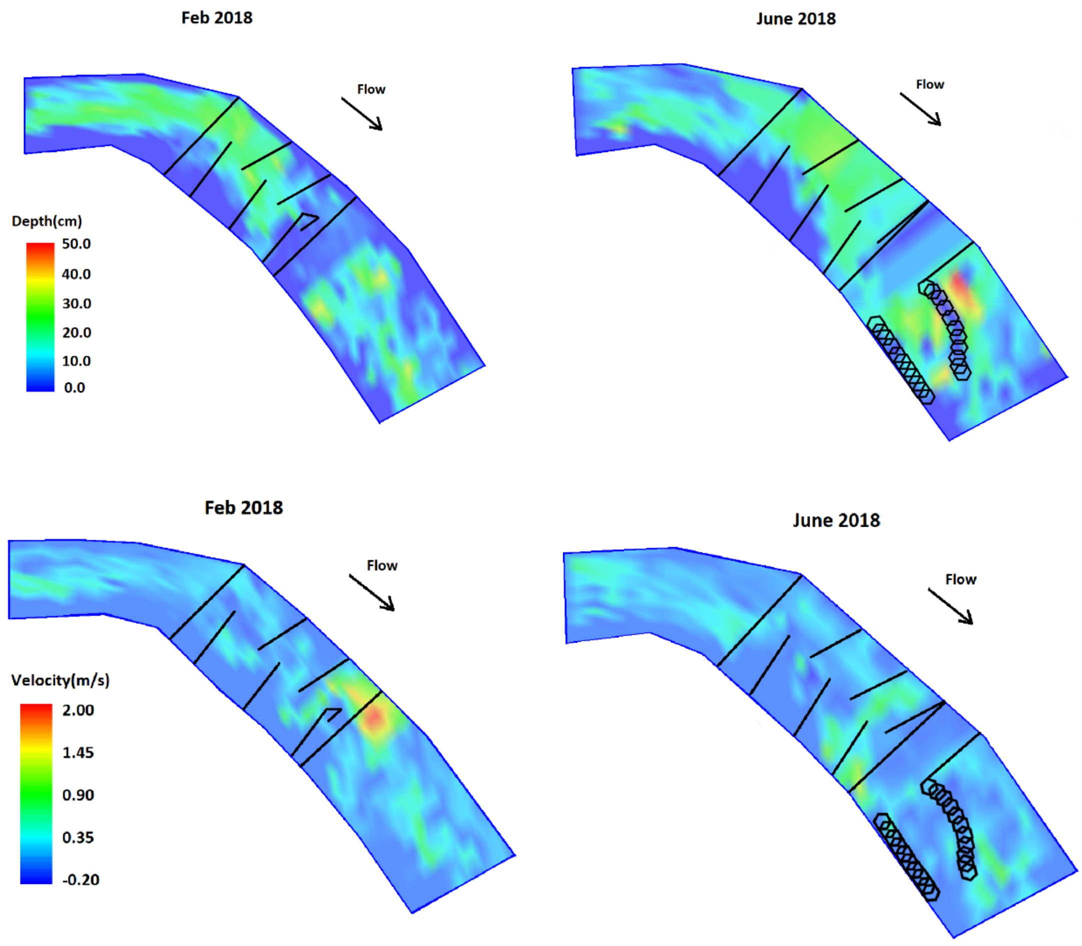


Figure s5.3 Water depth and flow velocity (measured at half depth) before and after the baffles re-construction and rock ramp installation at Pelton Fell Bridge (top: depth; bottom: velocity).

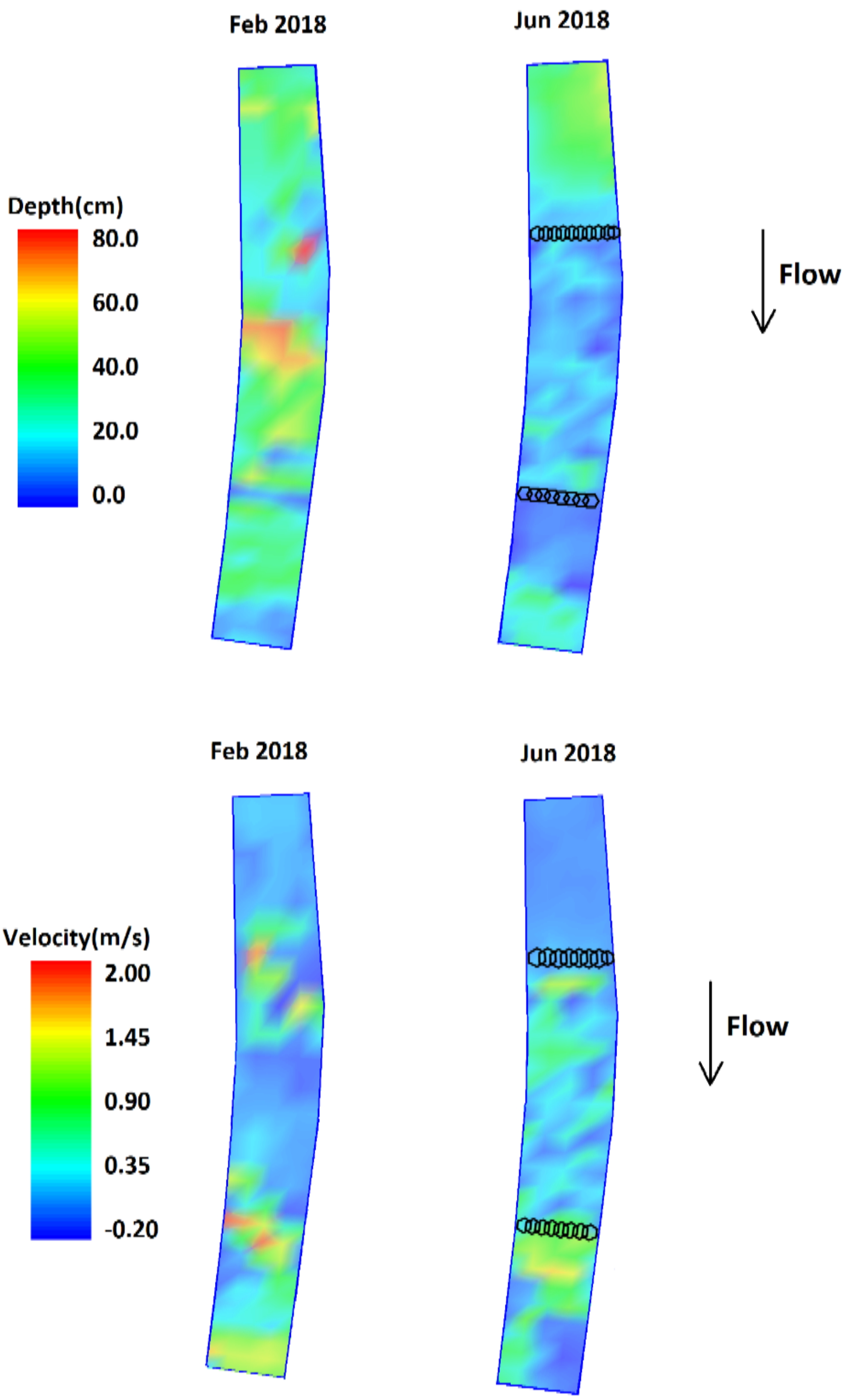


Figure s5.4 Water depth and flow velocity (measured at half depth) before and after the rock ramp installation at Newfield Bridge (top: depth; bottom: velocity).

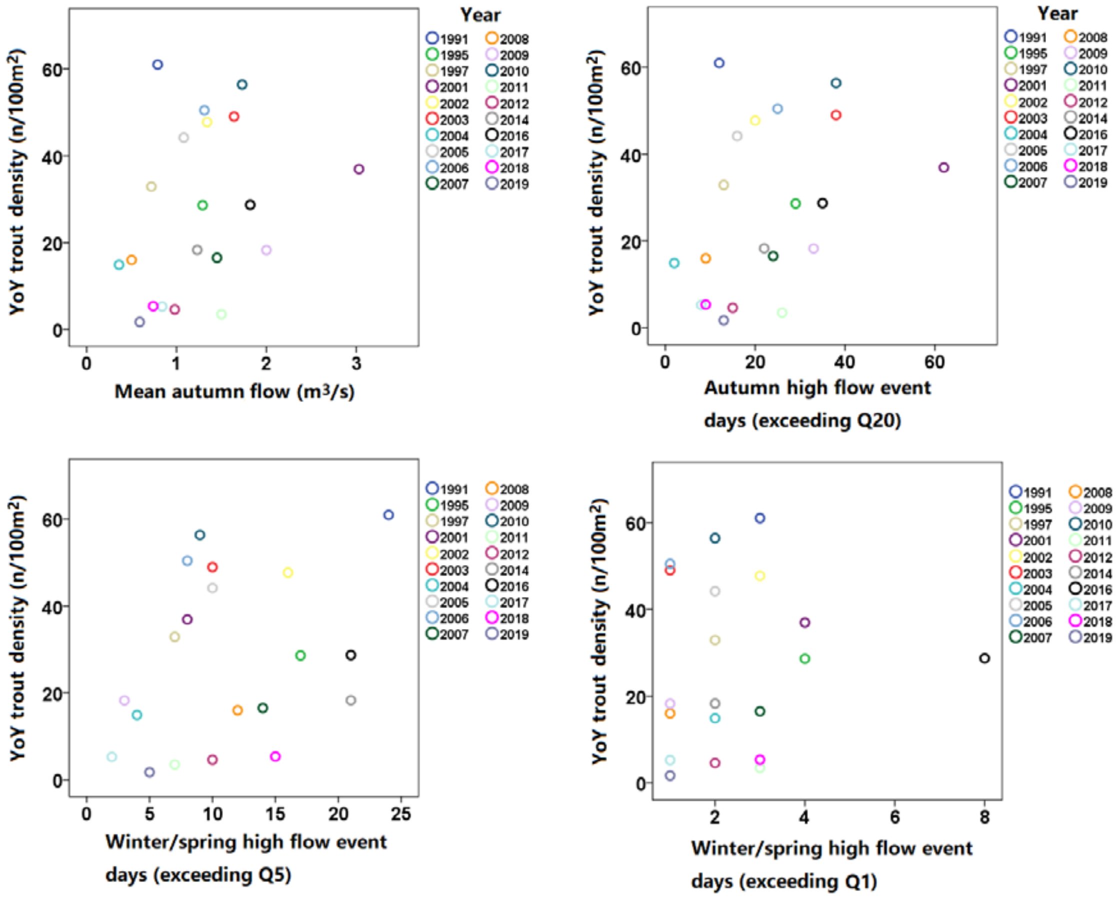


Figure s5.5 Relationship of YoY trout minimum density and mean daily flow / high flow event days in the Bedburn Beck between 1991 and 2019.

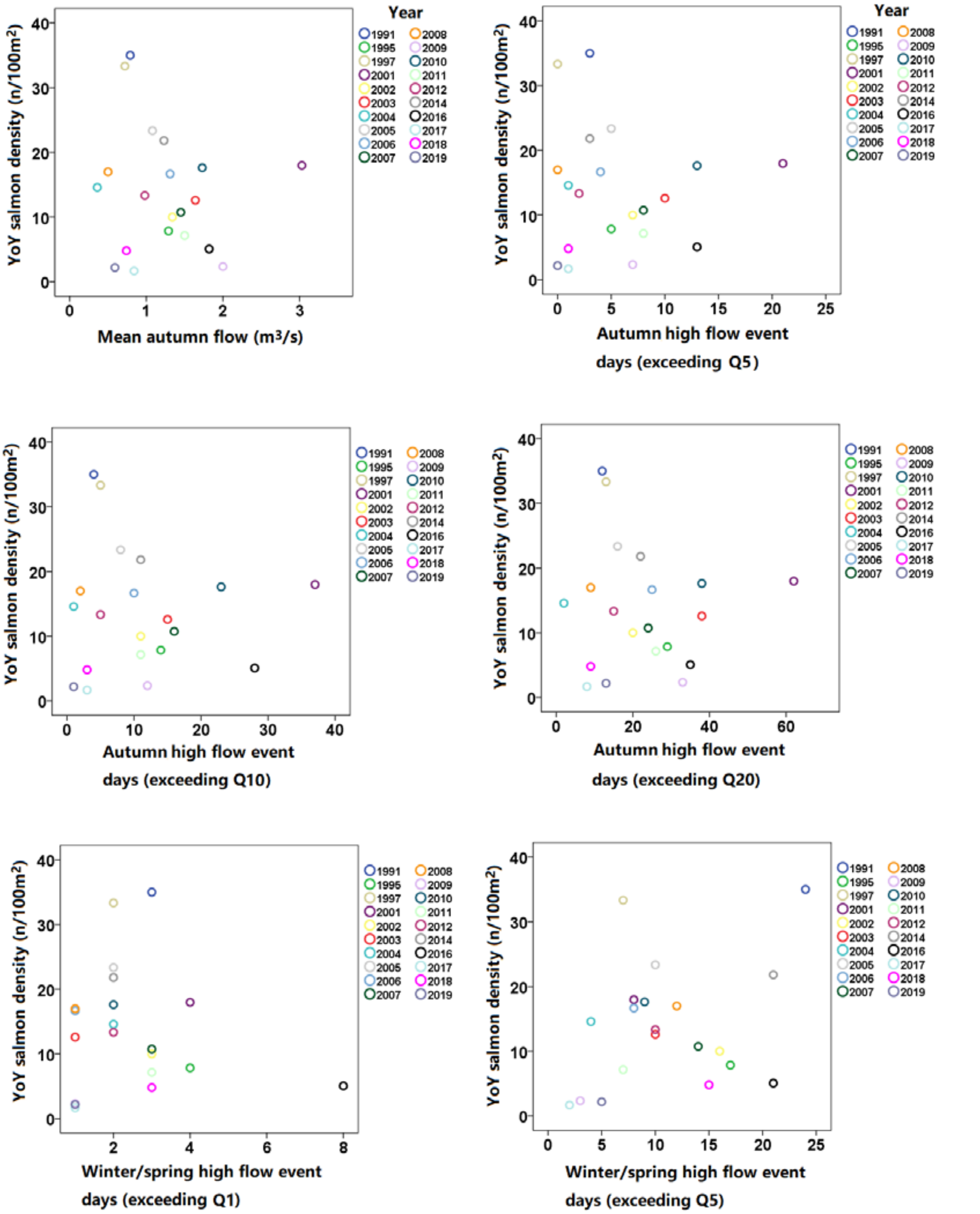


Figure s5.6 Relationship of YoY salmon density and mean daily flow / high flow event days in the Bedburn Beck between 1991 and 2019.

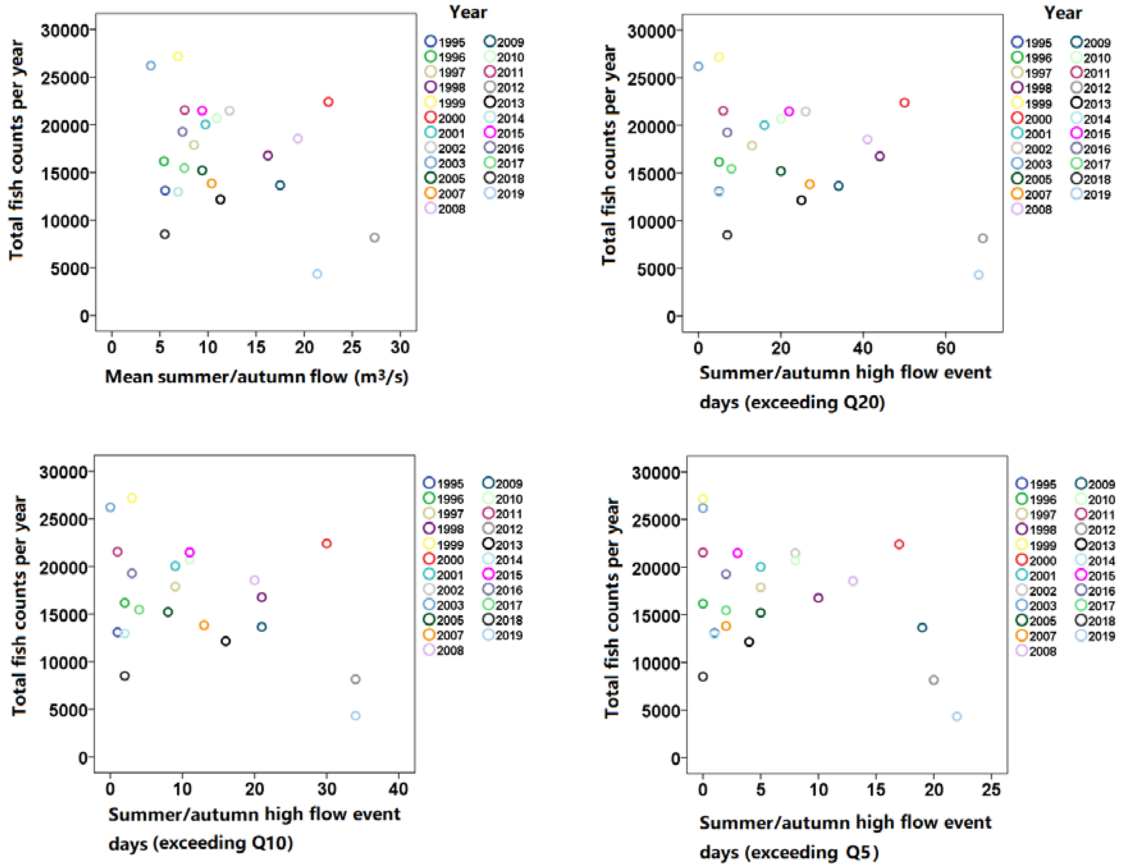


Figure s5.7 Relationship between Framwellgate annual fish counter data and mean flow at Chester-Le-Street gauge station from 1 June to 30 November, as well as numbers of high flow event days (exceeding Q5 or Q10 or Q20).

Table s5.1 Capture efficiency (%) of YoY trout, Older trout, bullhead and minnow by three-pass electro-fishing and single-pass method in River Deerness. Data calcluated using the proporation of the estimated fish population (Carle and Strub, 1978) that captured. Single-pass data was the first run data extracted from three-pass method.

Method	Year	Species	Mean	SE	Range
Three-pass	2016	YoY	89.03	1.96	77.27-100
		Older	96.88	1.24	84.62-100
		BH	89.78	1.47	79.31-100
		MN	94.53	1.58	81.25-100
	2017	YoY	92.76	1.96	79.07-100
		Older	91.86	2.67	61.21-100
		BH	89.82	2.54	69.23-100
		MN	94.94	2.42	72.73-100
	2018	YoY	88.65	3.27	56.06-100
		Older	96.38	1.41	81.48-100
		BH	81.14	4.21	39.02-100
		MN	82.04	6.17	56.06-100
	2019	YoY	87.43	3.96	42.67-100
		Older	95.21	2.55	68.42-100
		BH	78.64	2.85	61.68-97.06
		MN	86.1	3.14	57.14-100
Single-pass	2016	YoY	49.66	3.39	29.63-72.73
		Older	64.55	4.03	38.46-87.5
		BH	49.03	1.87	37.5-64.71
		MN	57.9	4.15	31.25-100
	2017	YoY	56.37	5.13	0-84.21
		Older	58.13	4.49	22.42-88.24
		BH	50.9	4.43	28.85-100
		MN	67.44	7.06	18.18-100
	2018	YoY	50.14	4.73	21.21-80
		Older	67.72	4.25	37.04-100
		BH	44.68	5.67	4.88-100
		MN	34.07	7.66	0-72.73
	2019	YoY	52.36	4.36	12-75
		Older	66.85	4.93	26.32-100
		BH	41.06	3.24	25.23-64.71
		MN	33.18	5.34	0-54.9

Table s5.2 Mann-Whitney U Test showing the differences in trout density between the paired sites after the connectivity restoration in the River Deerness.

Sites	Barrier	Restoration time	Method	Total	YoY	Older
S1 - S2	B2	Oct 2013	Rock ramp	U = 12 P = 0.337	U = 8 P = 0.109	U = 16.5 P = 0.810
S3 - S4	B3	Oct 2013	Natural bypass	U = 10 P = 0.2	U = 4 **P = 0.025**	U = 11.5 P = 0.297
S5 - S6	B4	N/A	N/A	U = 23 P = 0.848	U = 24 P = 0.949	U = 13 P = 0.141
S7 - S8	B5	Apr 2014	Removed	U = 17 P = 0.873	U = 14 P = 0.522	U = 15 P = 0.631
S9 - S10	B6	Apr 2014	Removed	U = 12 P = 0.337	U = 13 P = 0.423	U = 17 P = 0.873
S11 - S12	B7	Aug 2014	Removed	U = 8 P = 0.347	U = 7 P = 0.251	U = 10 P = 0.602
S13 - S14	B8	N/A	N/A	U = 24 P = 0.949	U = 20.5 P = 0.609	U = 23 P = 0.848
S15 - S16	B9	Oct 2012	Rock easement	U = 17 P = 0.338	U = 18 P = 0.406	U = 19 P = 0.482

Table s5.3 Capture efficiency (%) of YoY trout, Older trout, bullhead and minnow by three-pass electro-fishing and single-pass method in Brancepth Beck. Data calcluated using the proporation of the estimated fish population (Carle and Strub, 1978) that captured. Single-pass data was the first run data extracted from three-pass method.

Method	Year	Species	Mean	SE	Range
Three-pass	2017	YoY	93.24	1.25	85.84-100
		Older	99.66	0.34	92.86-100
		BH	89.42	2.19	68.18-100
		MN	79.12	6.6	72.52-85.71
	2018	YoY	89.23	2.63	61.17-100
		Older	97.44	1.06	80.56-100
		BH	82.6	3.99	46.15-100
		MN	95.74	4.26	82.95-100
	2019	YoY	93.93	2.16	72.76-100
		Older	99.09	0.65	88.89-100
		BH	95.06	1.96	66.67-100
		MN	85.97	2.38	78.26-91.23
Single-pass	2017	YoY	52.34	3.39	0-68.25
		Older	74.46	4.79	33.33-100
		BH	46.59	3.32	20.45-75
		MN	37.89	7.35	30.53-45.24
	2018	YoY	52.1	4.85	24.74-100
		Older	59.44	6.94	0-100
		BH	42.18	6.28	0-100
		MN	68.4	7.65	50.39-85.71
	2019	YoY	58.32	4.27	33.33-100
		Older	74.79	6.32	20-100
		BH	59.55	5.82	0-100
		MN	44.11	3.48	32.61-54.39

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